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Discover Soil

**REVIEW** **Open Access**

# Management and valorisation of sewage sludge to foster the circular economy in the agricultural sector

Caterina Lucia[1], Luigi Badalucco[1], Santo Fabio Corsino[2*], Antonino Galati[1], Massimo Iovino[1], Sofia Maria Muscarella[1], Sara Paliaga[1], Michele Torregrossa[2] and Vito Armando Laudicina[1,3]

*Correspondence:
Santo Fabio Corsino
santofabio.corsino@unipa.it
[1] Department of Agriculture, Food and Forest Sciences, University of Palermo, Viale delle Scienze, building 4, Palermo 90128, Italy
[2] Department of Engineering, University of Palermo, University of Palermo, Viale delle Scienze, building 8, Palermo 90128, Italy
[3] NBFC, National Biodiversity Future Center, Palermo 90133, Italy

**Abstract**
Sewage sludge, a by-product of wastewater treatment, is a potential source of energy and resources. Its use as raw material presents a promising perspective for waste management within the framework of the circular economy. Despite its potential benefits, the excessive production of sewage sludge poses environmental and socio-economic challenges, including the threat of contamination by heavy metals, pathogens and organic micro pollutants. Moreover, the European Directive regulating the agricultural use of sewage sludge (86/278/EEC) does not fully reflect current scientific knowledge and technological advancements, particularly in regard to emerging contaminants and harmonised reuse strategies among Member States. To date, only a limited number of comprehensive reviews have addressed the main impacts of sewage sludge in the agricultural sector, particularly regarding its effects on soil physical, chemical, and biological properties. Sewage sludge valorisation, including soil fertilization, plays a pivotal role in improving soil quality and long-term productivity. This is proven by an increased supply of organic matter and nutrients, as well as improvements in soil ecosystem health that promote crop growth. In addition, the application of sewage sludge makes the soil structure more stable and less vulnerable to erosion. Furthermore, through different physical-chemical processes, it is possible to recover materials and energy to be used as end products or included in other production processes, thus contributing to promote a sustainable and circular economy approach. The implications of this review point to the need for suitable EU laws and regulations, greater social acceptance, and continued research to exploit the full potential of sewage sludge for agricultural purposes.

## 1 Introduction

Wastewater generation is a massive global issue that results from several human activities and industries, posing major obstacles to the sustainable management of water resources [1]. Wastewater production has increased dramatically because of the rapid growth in population, urbanization, and industrialization, posing a severe threat to human health and the environment [2]. According to Jones et al. [3]global wastewater production has been estimated to be about $360 \times 10^9$ m [3] yr$^{-1}$, of which 63% is collected and 52% is, in turn, treated. Indeed, contaminants present in wastewater are successfully removed to comply with environmental regulation, before wastewater is released into water bodies or used for other purposes, thus protecting both human health and the environment. The major constraint in wastewater treatment operations is the sludge production [4]. Sludge is produced as a result of the physical, chemical, and biological processes, which aim to remove pollutants present in soluble and suspended form from wastewaters [5]. Directive 86/278/EEC, regulating the protection of the environment, and in particular of the soil, when sewage sludge (SS) is used in agriculture, defines sludge as: "*(i) residual sludge from sewage plants treating domestic or urban waste waters and from other sewage plants treating waste waters of a composition similar to domestic and urban waste waters; (ii) residual sludge from septic tanks and other similar installations for the treatment of sewage; (iii) residual sludge from sewage plants other than those referred to in (i) and (ii)*".

In this review, the term *sewage sludge (SS)* is used in accordance with the terminology adopted in European regulations. It refers to the residual semi-solid material generated during wastewater treatment. However, it is important to note that in other contexts, particularly in North America, the term *biosolids* is commonly used to indicate treated sewage sludge that meets specific regulatory standards for safe land application, including pathogen and contaminant reduction [6]. Urban, industrial and agricultural wastewaters are just a few of the sources that might produce sewage sludge. Despite being often considered a by-product, sewage sludge possesses the potential to serve as a sustainable source of energy and/or resources in the view of the circular economy approach. This capability enables the substitution of a corresponding quantity of materials or energy that would otherwise be derived from non-renewable resources, carrying significant environmental advantages [1]. The utilization of sludges as a raw Material in various industries presents a promising perspective for waste Management within the framework of the circular economy. This because the organic components within sludge contain valuable energy and nutrient resources. A study conducted in 2015 by the International Solid Waste Association (ISWA) revealed a significant advantage in the context of the circular economy. It highlights that the energy and fuels derived from waste not only offer a sustainable alternative but also have the potential to substitute other energy sources, thereby mitigating concurrent $CO_2$ emissions [1]. The composition of sewage sludge fluctuates widely, depending on the sources and treatment methods. There are many techniques to recover and use the considerable amounts of organic matter, nutrients (such as nitrogen and phosphorus), and energy that can be found in sludges [7, 8]. Indeed, sludge management handled properly can support sustainable agricultural practices, energy production, and resource conservation [7].

Several researchers have shown that the application of sludge in agriculture provides improvements in soil fertility and consequently in crop yield [9–11]. The application of SS as agricultural practice offers a potential reduction in the need for conventional fertiliser production [7]. Specifically, from 2017 to the end of 2024, global demand of agricultural inorganic fertilizers is expected to increase at an average annual rate of 0.7%, to reach 197.1 Mt by nutrients (N, $P_2O_5$, $K_2O$) [12]. Such an increase is expected to grow even more in the future. However, sludge production could pose risks and challenges if it is not properly managed, due to the possible content of heavy metals, contaminants, and harmful pathogens [13]. It has been reported that the United States is generating 40 Mt of sludge annually, while the European Union produces about 50 Mt of sludge. Among the European country, Germany is the highest sludge producer, followed by the United Kingdom and France. Moreover, it has been estimated that these countries together with Spain and Italy generate about 75% of the European sludge [14]. Therefore, different management strategies and sludge treatment technologies have been developed to solve the challenges related to sludge accumulation [15]. These consist of techniques including anaerobic digestion, composting, thermal drying, and incineration, all working to reduce sludge volume, stabilize its organic content, and improve the recovery of valuable materials [16, 17].

Selecting a sustainable treatment technology for sewage sludge must match environmentally friendly, cost-effective, and socially acceptable criteria. The global industry faces various challenges and risks, such as a fluctuating commodity market, difficulties in cost management, restricted access to financing, and increasing demands for social, economic, and environmental engagement from host communities and regulatory bodies. Consequently, solutions should not only address the challenges and issues associated with sludge disposal for energy production and nutrient recovery but also contribute to a sustainable environment by reducing harmful substances while simultaneously supporting economic benefits [18].

The keyword co-occurrence Map provides a comprehensive overview of Major research trends related to sewage sludge from 1954 to 2024 (Fig. 1). The centrality of the term "sewage sludge" reflects its foundational role in the field. Closely associated keywords such as "heavy metals," "soil," and "compost" indicate a strong research focus on environmental applications, particularly in agriculture, and on contaminant-related issues. Additional frequent terms, such as "anaerobic digestion," "phosphorus," "pathogens," and "microplastics", point to growing interest in resource recovery, health risks, and emerging pollutants. Meanwhile, the peripheral positioning of terms like "gasification," "biogas," and "life cycle assessment" suggests increasing attention to energy recovery and sustainability. This map highlights the multidisciplinary nature of sewage sludge research, spanning environmental science, agriculture, waste management, and public health.

Accordingly, this review aims to provide a comprehensive overview of sludge production, examining its complexities, challenges, and potential. Particular emphasis is placed on the most effective technologies for sludge treatment, resource recovery, and beneficial reuse. The main focus, however, is the valorisation of sewage sludge in agriculture and its effects on soil properties (physical, chemical, and biochemical). Unlike previous reviews, this work also addresses the economic opportunities and barriers associated with sludge reuse. Ultimately, the goal is to support the development of effective and integrated policy frameworks that align with circular economy principles.

This review contributes to the substantive enrichment of the literature on the current management and valorisation of sewage sludge in agriculture. Compared to the current literature, this review focuses on the valorisation of sewage sludge in agriculture, especially its effects on soil physical, chemical, and biological properties. Other topics, such as energy and material recovery, economic and regulatory implications and mentioned only as part of a circular economy context. This represents a significant improvement for

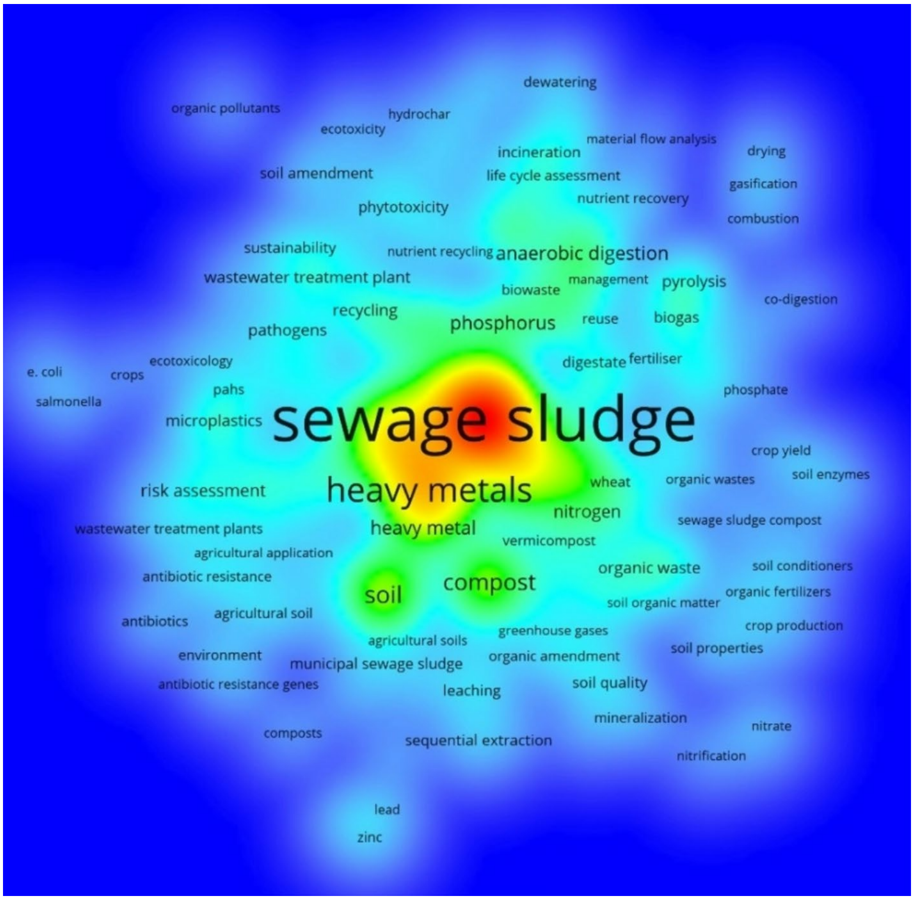

**Fig. 1** Keywords co-occurrence Map from the Scopus database. This map is based on Literature types, such as papers and review papers, covering the publication period from 1954 to 2024. The keyword co-occurrence count includes 530 keywords with a frequency greater than five within the database

the scientific and technical-legislation knowledge about this topic, aiming at providing a comprehensive and updated state-of-the-art of all the key aspects related to sewage sludge management.

## 2 Sludge generation and composition

Sewage sludge is generated in municipal wastewater treatment plants (WWTPs) at different stages of the wastewater purification process. Indeed, in WWTPs based on the activated sludge process, the sewage sludge can be classified into two types. The first is named primary sludge, which is originated from the initial physical separation of settleable solids in the primary settlers. It is characterized by high putrescibility with a total solids (TS) content ranging from 2 to 7%. The second is the secondary sludge, that results from bacterial growth, decay phenomena and is separated from the treated effluent in the final settlers. Secondary sludge contains microorganisms, endogenous residue, and inert solids not removed in the primary settling or entering with the raw wastewater. The TS content in secondary sludge ranges from 0.5 to 1.5%. Both primary and secondary sludge are mixed and sent to the sludge handling unit for thickening, digestion and dewatering processes [19]. Overall, the sewage sludge produced by a WWTP is variable and it depends on the plant potential, the technology adopted for biological process and operating conditions. As an order of Magnitude, the specific sludge production in

**Table 1** Main chemical properties of urban sewage sludge

| Parameters | Value | References |
|---|---|---|
| pH | 7.4 | Baghina et al. [23] |
| | 6.3 | Consuegra et al. [24] |
| | 7.7 | Hamdi et al. [26] |
| | 7.1 | Ahmad et al. [27] |
| | 8.0 | Achkir et al. [28] |
| Electrical Conductivity (dS $m^{-1}$) | 4.3 | Consuegra et al. [24] |
| | 1.7 | Hamdi et al. [26] |
| | 2.5 | Ahmad et al. [27] |
| | 1.7 | Achkir et al. [28] |
| Organic matter (%) | 64 | Baghina et al. [23] |
| | 34 | Consuegra et al. [24] |
| | 65 | Achkir et al. [28] |
| Total organic C (g $kg^{-1}$) | 185 | Hamdi et al. [26] |
| | 365 | Ahmad et al. [27] |
| Total N (%) | 1.9 | Baghina et al. [23] |
| | 4.1 | Consuegra et al. [24] |
| | 1.2 | Hamdi et al. [26] |
| | 3.7 | Achkir et al. [28] |
| Total P (g $kg^{-1}$) | 0.1 | Baghina et al. [23] |
| | 19.9 | Consuegra et al. [24] |
| K (g $kg^{-1}$) | 2.3 | Baghina et al. [23] |
| | 4.3 | Consuegra et al. [24] |
| | 0.1 | Hamdi et al. [26] |
| | 0.2 | Achkir et al. [28] |

**Table 2** Main European Directives and Regulations on sludge management and disposal

| Directives and regulations | Aims |
|---|---|
| Directive 86/278/EEC | Aims to regulate sewage sludge use in agriculture to prevent harm to soil, plants, animals, and humans. |
| Directive 91/271/EEC | Seeks to protect the environment from urban and industrial wastewater discharges. |
| Directive 91/676/EEC | Seeks to protect the environment from urban and industrial wastewater discharges |
| Directive 2008/98/EC | Establishes a legal framework for waste management to protect health and the environment, promoting recycling and reuse. |
| Directive 2018/851/EC | Enhances waste management laws to support the transition to a circular economy, focusing on waste prevention and management. |
| Regulation 2019/1009 | Sets rules for EU fertilizing products to ensure safety, quality, environmental protection, and market consistency. |

*dm*: dry matter

WWTPs varies widely from 35 to 85 g dry solids per population equivalent (PE) per day (gTS $PE^{-1}$ $d^{-1}$) [20].

The content of volatile, i.e. organic, solids in the sludge ranges between 70 and 80% before the stabilization process, whereas after sludge digestion, it decreases to about 40–50%, thus reducing its putrescibility [21]. Similarly, the water content is highly variable. Indeed, it ranges from 98 to 99% in the sludge before stabilization, and it reduces up to 70–80% after dewatering with conventional mechanical processes. In some cases, the water content could be further decreased by advanced drying treatment, leading to a reduction in sludge moisture to less than 10–20% to reduce the cost of sludge disposal.

Besides oxygen and hydrogen, main elements of sewage sludge are carbon, nitrogen, and phosphorous (Table 1). It was estimated that one ton of dry sewage sludge contains on average 8 kg of P, 6 kg of N, 200 kg of organic matter [22]. Besides, it contains a wide variety of mineral elements, like magnesium, calcium, and potassium, as soluble salt, iron, and aluminium. In addition, several trace elements can be found, such as As, Cr, Mo, Ni, Pb, Sn, which make it a potential resource for mineral recovery or for soil pollution.

However, recent studies found several emerging contaminants, such as pharmaceutical compounds, microplastics (MPs), and antibiotics in sewage sludge samples [29].

Microplastics, whose sizes range from 1 μm to 5 mm, were found in several sewage sludge samples all over the world [29]. Their abundance is highly variable ranging between 30 and $100 \times 10^3$ particles/kg dry sludge. The main shape of MPs observed belongs to fragments and polyethylene is the most prevalent. Their presence in sludge could be detrimental for several applications, since MPs could be harmful for microorganisms and the environment in general.

A recent study revealed the presence of different types of emerging contaminants in sewage sludge from all over the world. Specifically, pharmaceuticals and personal care products (PPCPs), among which triclosan, ciprofloxacin, and diclofenac, were found in concentrations ranging from 0.1 to 160 $\mu g\ kg^{-1}$. In addition, flame retardants (e.g., polybrominated diphenylethers, decabromodiphenyl ethane), perfluorinated alkylated substances (PFAS), hydrocarbons and surfactants are omnipresent and pose risks to human health and the environment [30]. Consequently, using sludge directly for agricultural purposes poses significant risks. Among the environmental factors influencing the behaviour of heavy metals in sewage sludge–amended soils, soil pH plays a critical role. Acidic conditions increase the solubility of metals such as cadmium and zinc, enhancing their mobility and uptake by plants, while alkaline pH reduces solubility through precipitation or adsorption processes.

Furthermore, not only the total concentration but also the speciation, that is, the chemical form in which a metal is present, determines its mobility, bioavailability, and ecotoxicity. Recent findings by Feng et al. (2023) [31] reported that over 70% of Zn and Ni in sewage sludge were present in exchangeable or acid-soluble fractions, posing a higher environmental risk compared to more stable forms such as those bound to organic matter or mineral phases. In contrast, lead and chromium tend to occur in less mobile forms, while cadmium remains more available, especially in acidic soils.

These results suggest that amendments capable of increasing soil pH, such as biochar, can effectively immobilize labile metal fractions and thereby reduce their potential toxicity in agricultural applications.

## 3 European sewage sludge directive

The Management and disposal of sewage sludge is a key issue in relation to the risk of secondary pollution resulting from its mismanagement. European legislation on sewage sludge mainly aims to regulate its treatment and disposal to protect Human Health and the environment. Over the last 30 years, the EU has extensively regulated the management and reuse of sludge with various legislative instruments and acts, as this is part of the EU vision of sustainability for an environmentally safe approach, also in relation to the rapid growth in the amount of sludge produced [10].

The most significant European directives related to the Management and reuse of sludge in agriculture is the Directive 86/278/EEC. These are complemented by the public consultation [32] launched in 2020 by the EU with the aim of renewing Directive 86/278/EEC (Table 2).

The Sewage Sludge Directive (86/278/EEC, 1986) sets up strict guidelines on the use of sewage sludge as a fertiliser in agriculture, in order to avoid harmful effects on the environment and human health. It also considers the nutrient needs of plants, without compromising the quality of the soil and surface or groundwater [10]. This directive imposes limit values for heavy metals content in both sewage sludge and soil to which it is applied. The directive establishes maximum permissible concentrations for contaminants such as cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), zinc (Zn), mercury (Hg) and chromium (Cr), to prevent their accumulation in agricultural soils. Furthermore, the Sewage Sludge Directive imposes restrictions on the application, specifying that untreated sludge must not be used on land where crops are being cultivated or will be harvested within a certain period. The main points of the Sewage Sludge Directive require that sludge must undergo a stabilisation process, such as composting, before being used in agriculture. However, in some EU countries (e.g. Germany, Denmark, Netherlands), farmers may be allowed to use untreated sludge if it is injected or buried in the soil.

Sewage Sludge Directive, in detail, contains three Annexes: Annex IA reports the limit values for heavy metals in the soil, Annex IB regulates the maximum concentration of heavy metals in sludge and Annex IC the maximum annual amounts of heavy metals that may be released into the soil (Table 3).

However, the Sewage Sludge Directive does not restrict severely limits on Maximum content values for heavy metals, which is why EU countries have introduced stricter Limits for the use of sludge in agriculture. Indeed, out of 27 countries, 18 have introduced restrictions on cadmium (Cd), 14 on copper (Cu), 19 on mercury (Hg), 16 on nickel (Ni), 14 on lead (Pb), 10 on zinc (Zn) [11, 33].

Sewage Sludge Directive [33] contains no limit values or special requirements for organic micropollutants. Recent developments have reviewed the European Union Directive and update its sewage sludge policies to address emerging concerns and improve sustainability. Several States have added limit values in their national regulations. The organic compounds most controlled by various States are polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs) and organic absorbable halogens (AOX) [11]. A critical limitation of the Sewage Sludge Directive (86/278/EEC) lies in its inability to fully address current industry needs, given the advancements in scientific understanding and the emergence of new pollutants such as pharmaceuticals, microplastics, and persistent organic contaminants. Recent reviews emphasise that an update is necessary to ensure environmental safety and coherence with the circular economy principles [1, 34]. This is why, in 2020, the European Union created an initiative to encourage citizens, stakeholders and plant operators at European level to express their opinion on the reformulation and amendment of the Sewage Sludge Directive [35].

**Table 3** Limit values given in Annex IA, Annex IB, and Annex IC of directive 86/278/EEC

| Parameters | Cd | Cu | Ni | Pb | Zn | Hg | Cr |
|---|---|---|---|---|---|---|---|
| Annex IA (mg kg$^{-1}$) | 1–3 | 50–140 | 30–75 | 50–300 | 150–300 | 1-1.5 | - |
| Annex IB (mg kg$^{-1}$ of d.m.) | 20–40 | 1000–1750 | 300–400 | 750–1200 | 2500–4000 | 16–25 | - |
| Annex IC (kg ha$^{-1}$ y$^{-1}$) | 0.15 | 12 | 3 | 15 | 30 | 0.1 | - |

*dm*: dry matter

Subsequently, a "Public Consultation" was launched in 2021 to collect the opinions of stakeholders, operators, and experts in the field at European level, in order to modify the Directive in light of new technologies.

## 4 Sludge management/disposal

Sewage sludge management represents one of the greatest challenges to be addressed for maximising the efficiency of wastewater treatment plants [36]. Indeed, the primary constraint is related to the sludge volume reduction and minimization of management and transportation costs, due to the presence of significant levels of moisture that make dewatering operations critical [35]. Since they contain 97–98% of water, it would be advisable to treat them properly in order to adopt the correct management strategy [36]. Recently, several processes have been developed aimed, on the one hand, at achieving higher dehydration levels than traditional ones based on mechanical dehydration, and on the other hand at promoting a reduction in sludge production directly in the biological compartment. Among the former, several types of thermal drying systems have been proposed, at high temperatures (120° – 180°) based on convective, conductive, and combined systems, but also systems based on the exploitation of solar energy to dry the sludge at low temperatures inside greenhouses prepared for this purpose, and drainage systems in geotextile tubes [35]. The processes that implement the reduction of excess sludge production at the source are based on different biochemical mechanisms: cell lysis-cryptic growth, decoupled metabolism, endogenous metabolism and high-temperature oxidation. However, despite being able to achieve reductions of up to 70–80%, these processes are not yet widely adopted. The most appropriate strategy to be adopted for sewage sludge treatment depends not only on the geographical location and socio-economic situations, but also on the technical costs and all environmental regulations [37, 38].

Sewage sludge disposal can be performed through different technical operations including sanitary landfill, incineration, composting and anaerobic digestion. Depending on the adopted methods, different environmental impacts may occur. According to Eurostat (2022) [39]the two primary approaches to Managing sewage sludge in the EU are incineration and agricultural use. Although the EU employs 40% of its total sludge production for agricultural purposes, the amount of sewage sludge that is applied to the soil varies greatly throughout the member States. In some EU countries (Germany, France, the United Kingdom and Portugal) the 30–70% of the produced sewage sludge is utilized for agricultural purposes, while in Belgium, Holland, and Sweden, its application is rarely practiced [40]. Incineration procedure not only reduces the sewage sludge volume (about 90% of the entire volume), but it can also significantly reduce the dangers to public health that may arise from pathogens. Moreover, incineration does not require a long-term reservation or long-distance transportation compared to composting or landfilling [41].

Waste management represents an ongoing challenge, especially for countries with limited resources. This difficulty arises primarily from inconsistent legislation, the absence of a systematic approach to identify appropriate sludge management strategies, and the high costs involved in modernizing and investing in obsolete sewage treatment plants.

## 5 Valorisation of sludge

The valorisation of sludge through different methods represents a sustainable approach to waste management (Fig. 2). In the following sections, the most relevant aspects of sewage sludge valorisation related to soil fertilisation, composting and energy recovery will be discussed in detail. More specifically: (i) it will be discussed how sludge acts as a nutrient-rich organic amendment, enhancing soil fertility and soil structure; (ii) it will be examined how sludge composting transforms organic sludge into nutrient-rich compost suitable for soil conditioning and plant growth; and finally, (iii) it will be explored how to convert organic matter held in sludge into biogas through processes like anaerobic digestion. The valorisation of sludge not only minimizes the environmental impact of waste disposal but also promotes resource efficiency, supporting the circular economy.

### 5.1 Soil fertilization

Sludge valorisation through soil fertilization represents a sustainable and efficient practice in waste management [42] and a "must" step in view of the shifting from a linear to a circular economy approach in the management of sludges. Sludge, being rich in organic matter and nutrients essential for plant growth, could be a valuable resource [43]. Indeed, the use of treated sewage sludge (SS) as a soil amendment improves soil structure, water retention, and nutrient availability, particularly nitrogen and phosphorus [44]. In Mediterranean regions, this practice helps counteract the decline in soil organic matter caused by high temperatures and intensive farming [45]. While some European countries reuse over half of their sludge in agriculture, others apply less than 5% or none at all [10]. According to Directive 86/278/EEC, sludge must undergo stabilisation (e.g., composting) before use. However, in certain EU countries (e.g., Germany, Denmark, Netherlands), untreated sludge may still be applied if injected or buried, which may benefit soil but also raises concerns about contaminants such as heavy metals, organic pollutants, and pathogens [45].

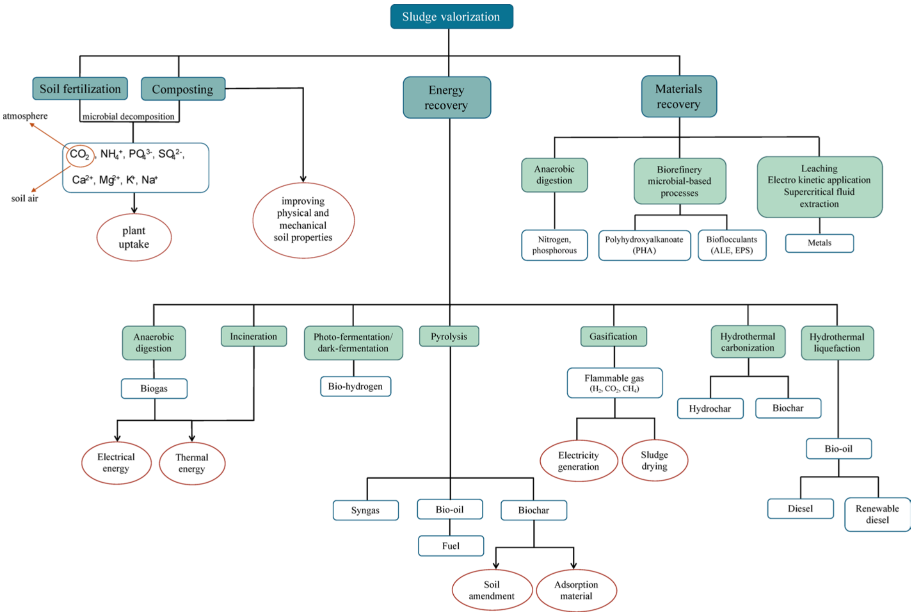

**Fig. 2** Scheme of the main products and applications resulting from sewage sludge valorisation

Indeed, such practice puts the emphasis on potential drawbacks concerning human health and the environment, due to the adverse effects of metals and pollutants such as pharmaceuticals that are often present in the sewage sludge [46–48]. These contaminants could leach throughout the soil profile, even reach the aquifer, if sewage sludge is used as fertilizer. However, sludge land application is highly encouraged to improve soil quality and reduce the demand for chemical fertilisers and overcome waste disposal problem.

The sewage sludge applied to the soil could replace to some extent chemical fertilisers. Nowadays, the harvest of natural resources is accompanied by the improper land application of fertilisers. Consequently, it would be advisable to estimate the correct amount of fertiliser to apply into a soil and to find alternatives, such as sewage sludge [49]. It has been estimated that total fertiliser consumption in terms of N, $P_2O_5$ and $K_2O$ supply is more than 183 million tonnes [50]. This data is expected to increase due to the increasing need for food production, which requires a regular supply of fertiliser [51]. Moreover, the prices of commercial fertilisers are destined to increase due to the large amount of energy required for their production [43].

Sewage sludge provides benefits as a fertiliser, especially in those areas where low-fertile soils cannot supply nutrients needed for plant growth. The addition of sludge as a recovery matter could offer a considerable improvement in terms of the supply of N, P, and K to plants and organic carbon in the soil, particularly in arid and semi-arid areas where organic matter is continuously decreasing [52]. There are several studies attesting the real efficiency of sludge as a soil improver. Jamil et al. [53] conducted a field experiment in Pakistan to study the effect of the addition of different levels of domestic sewage sludge on soil properties and yield of wheat crop. Sewage sludge, collected from the sewerage channels of Dera Ismail Khan City, air dried, ground and passed through a 4 mm sieve, were applied at a rate of 10, 20, 40, 60, 80 and 100 Mg $ha^{-1}$ to wheat crop. The incorporation of various levels of domestic sewage sludge into the soil resulted in a 29% increase of organic Matter and a significant increase in Macronutrients such as total N and available P by about 433%, while available K showed a 23% increase. Moreover, Jamil et al. [53] found that the application at a rate of 40 t $ha^{-1}$ increased spike length, number of productive tillers, number of grains spike, 1000-grain weight and grain yield. In another research by Hamdi et al. [26], a field-scale experiment was carried out in northeast Tunisia using municipal sewage sludge collected from a wastewater treatment plant. Sewage sludge was added to two different textured soils annually at a rate of 40, 80 and 120 t $ha^{-1}$. The supply of sewage sludge at a rate of 120 t $ha^{-1}$ increased TOC and the content of total N, available P and K by 2.92%, 0.18%, 246 and 22 mg $kg^{-1}$ in sandy loam soil and 2.64%, 0.15%, 356 and 19.2 mg $kg^{-1}$ in sandy soil, respectively. Findings similar to those of Jamil et al. [53] and Hamdi et al. [26] are reported by other authors (e.g [52, 54]). Generally, nearly 50% of the solid component of sewage sludge is organic matter, which significantly affects the physical, chemical and biological characteristics of the soil when applied. The presence of organic matter increases soil porosity, improving water retention and circulation. Moreover, some constituents of organic matter contribute significantly to soil aggregation, promote the decomposition of substances and establish a microbial balance [44]. Elsalam et al. [55] reported a positive effect on soil aggregate stability due to the addition of organic Matter by sludge compost with a bulk density of 0.75 g $cm^{-3}$.

Sustainable soil fertilization through sludge valorisation contributes to healthier and more productive soils, encouraging environmentally responsible agriculture while minimizing the need for synthetic fertilizers.

### 5.2 Biochar

Biochar is a porous product enriched in nutrients and mineral components obtained when biomass is transformed into a stable carbon enriched material by pyrolysis [56–58] that is a thermochemical process for the conversion of biomass into biochar, bio-oil, and syngas, generally performed at a temperature between 350 and 1000 °C in the absence or limited presence of oxygen [59, 60].

In recent years, more attention has been paid to the biochar obtained from sewage sludge as a soil amendment, in order to achieve an agronomic and environmental benefits. It improves soil nutrient and moisture retention capacity, microbial biomass, pH, organic matter, soil aggregate stability, and carbon sequestration [61–64]. For instance, Velli et al. [65] tested the effect of the application of sewage sludge biochar on soil properties. They produced biochar at a temperature of 300 °C and supplied it to the soil with or without compost at a rate of 2%. The results showed that soil organic C, mineral N, and available P increased by 67–85%, 55–145%, and 45–54%, respectively. Moreover, these positive effects were, in turn, confirmed by an increase in the dry weight of the stem and roots of tomatoes. The effect of biochar obtained from sewage sludge on soil properties depends on many factors, among which the temperature of pyrolysis. Ahmad et al. [27] investigated the effect of sewage sludge biochar, obtained at three different temperatures (300, 500, and 600 °C), on soil nutrients. After two months experiment, they found an increase in Ca, Mg, Na, Mn, and Zn concentration by increasing the pyrolysis temperature. Additionally, the maximum improvements in plant's yield were found at a pyrolysis temperature of 300 °C for both *Trigonella-foenum-graecum* L. and *Cicer arietinum* L.

Biochar from sewage sludge May have adverse effects on soil properties due to the presence of pollutants such as Heavy metals, possibly hazardous elements such as polycyclic aromatic hydrocarbons, and pathogens present in raw sewage sludge. These elements have the potential to contaminate soil, affecting the safety of people, animals, and plants. However, the pyrolysis process has several advantages because it generally reduces sludge volume by 80%, removes pathogens, and immobilizes metals into biochar thus reducing their leaching risk [66]. This may be ascribed to the higher ash content of the sewage sludge biochar, which would act as a basic buffer to prevent the release of heavy metals [67]. The amount of heavy metals that could accumulate depends on the heavy metals involved, the plants and the soil conditions [68]. Although the risk of heavy metals and other contaminants accumulation in sewage sludge biochar is mitigated by pyrolysis, there is a need to enhance the quality of these biochar to limit their harmful effects and improve their potential for agricultural purposes. Moreover, the cost and energy consumption required for sewage sludge pyrolysis must be considered while applying biochar. Indeed, the conversion of sewage sludge into biochar includes several processes such as feedstock supply, biochar pyrolysis, transportation, maintenance, storage, and even regeneration. Consequently, the development of stringent guidelines is required to achieve a biochar rich in nutrients with adequate metals content preserving environmental and economic benefits.

### 5.3 Composting

Due to its nutrients and organic matter content, sewage sludge could be applied to the soil as a conditioner. Unfortunately, sewage sludge can be harmful to the environment and humans due to the contaminants and pathogens it contains [69]. Therefore, the Sewage Sludge Directive stipulates that, before using, sewage sludge must undergo a stabilisation treatment, be suitable for producing a fertilising and/or soil-conditioning effect and not contain toxic, harmful, persistent, and/or bioaccumulative substances in dangerous concentrations to the soil, crops, animals, humans and the environment in general [70]. Consequently, for sewage sludge to be used for agricultural purposes, a high level of sanitisation and stabilisation of organic matter is required to maintain soil, water and air qualities, as well as to effectively utilise these bioresidues.

Composting is widely used and highly beneficial process for stabilizing the organic matter in sewage sludge [71]as it reduces toxic compounds and pathogens while producing a reusable product [72, 73]. The process involves three phases, that can be distinguished by the temperature and the microorganisms involved: mesophilic, thermophilic, and maturation [74]. In the mesophilic phase, microorganisms break down labile substrates, increasing temperature. The thermophilic phase, reaching up to 80 °C, eliminates pathogens and weed seeds, requiring sufficient oxygen to Maintain aerobic conditions. Once the temperature exceeds 70 °C, the compost is sanitised because pathogens are killed, weed seeds are inactivated and phytotoxic compounds are inactivated [11, 75]. Proper aeration, monitoring of key parameters, and maintaining optimal oxygen levels are essential for efficient composting and minimizing odors. Other fundamental control parameters include porosity, temperature, oxygen content, C/N ratio, moisture content, pH, electrical conductivity and cation exchange capacity [76]. To achieve successful composting, oxygen must be replenished through passive or forced aeration, or by turning the compost pile [77]. The optimal oxygen content in a well-ventilated compost pile is at least 5% during the active phase of composting (ideally closer to 10%). The occurrence of fermentation processes could slow down the composting process and producing unpleasant odours [78]. As nutrients deplete and microbial activity slows, the temperature drops, allowing mesophilic microorganisms to recolonize the pile. This Marks the cooling phase, During which the temperature gradually decreases to around 38°C [79, 80]. Finally, the maturation phase sees microbial succession and temperature decline, completing the composting process [11, 81].

It is important to understand how to assess the maturity or stability of the compost by evaluating several parameters (Fig. 3). The main ones are pH, C/N ratio, cation exchange capacity (CEC), ratio of humic substances (humic acids/fulvic acids, HA/FA) and electrical conductivity [82]. The pH must be between sub-alkaline and alkaline values, as acid pHs are characteristic of an unmatured compost. The C/N ratio is crucial, an optimal value should be between 10 and 15. It is strongly influenced by the initial biomass, but this value is taken as ideal as the C/N ratio in compost is comparable to the C/N ratio of Humic soil, which is close to 10. Another important parameter is the CEC, since it has been observed that it tends to increase during composting as organic materials are humified. Humification ratio (HA/FA) gives information about compost maturity because the HA/FA ratio increases during composting. Fulvic acids (FA) form first, then convert to humic acids (HA), resulting in stable, non-toxic humic substances. As compost matures, the concentration of humic compounds in the total organic matter increases to a stable

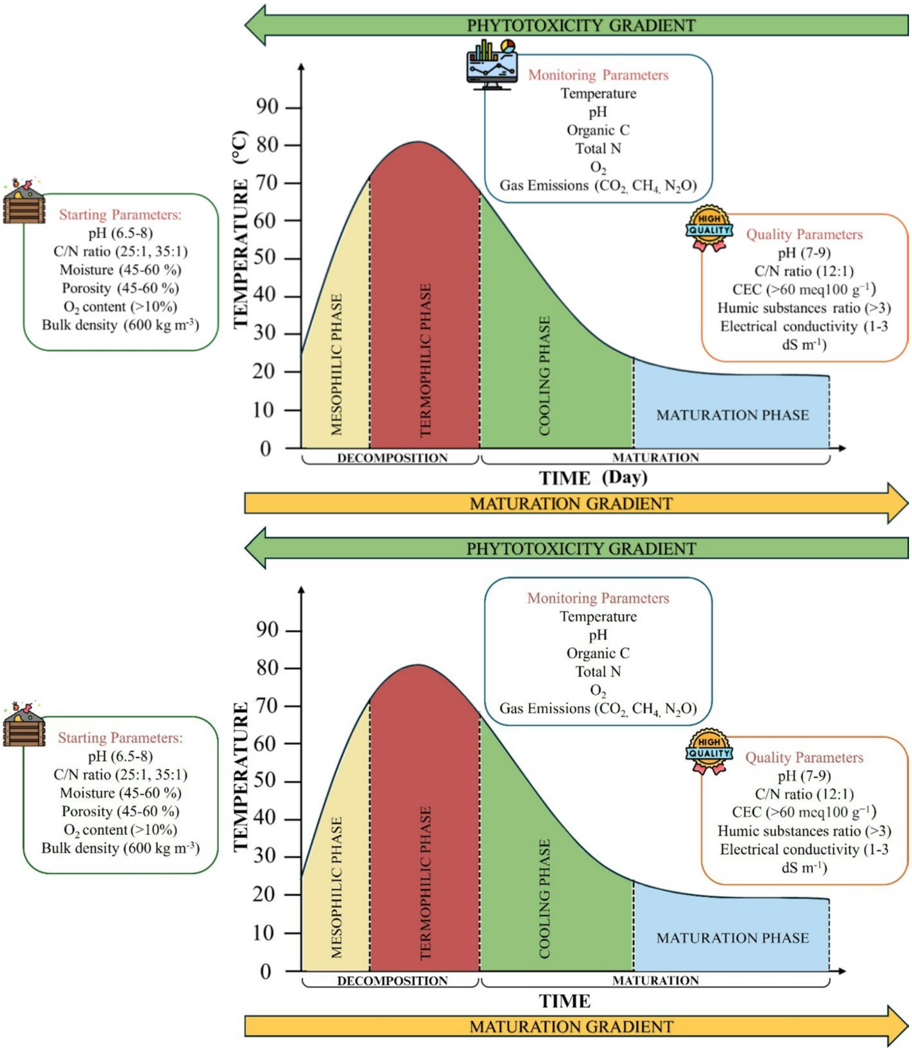

**Fig. 3** Composting process phases and parameters to control before, during, and after composting process

value. Finally, the electrical conductivity is crucial as the high salinity of the compost can be detrimental to the soil-plant system [82].

Sewage sludge must first undergo a water removal process before it can be composted. This step is essential for proper sludge treatment, as dewatering can minimise sludge volume, facilitate sludge transport, increase energy use efficiency, and even reduce leachate production in landfills [83]. However, it is difficult to dewater sludge directly due to the highly hydrated colloidal structures of microbial aggregates. The main methods of sludge dewatering, reported by Wu et al. [84]are divided into physical and chemical conditioning. Physical methods involve the addition of porous materials, sonication, heat treatment, freezing/thawing and electrical treatment. Chemical methods, on the other hand, involve coagulation or flocculation of the sludge particles, basic-acid treatments to break down the cell walls to release intracellular water, the advanced oxidation process and enzymatic treatments. Despite dewatering, sludge retains high level of moisture, which is why it is necessary to add a bulking agent to the composting mass in order to provide structural support, aerate the composting mass and improve the C/N ratio [85].

The bulking agent controls many parameters besides moisture content, such as temperature, dry matter, organic matter, pH, electrical conductivity, C/N ratio, greenhouse gas emission, mobility of heavy metals and the presence of other contaminants [11, 86]. The most commonly used bulking agents are wood chips, sawdust, hay and grass clippings, wheat straw, maize stalks, manure, biochar, agricultural waste and zeolite [87]. From the perspective of environmental sustainability and the circular economy, the choice of an appropriate bulking agent should consider not only its effect on the composting process and compost properties but also its origin, availability and proximity to the composting area. Different studies about the effect of the bulking agents on sewage sludge composting process are reported in Table 4.

The composted sewage sludge can be applied to the soil, significantly enhancing soil quality and boosting plant yield. For this reason, they have been extensively studied to understand how they can improve soil properties and agricultural productivity. Coutinho

**Table 4** State of the Art of effect of bulking agent on sewage sludge composting process

| **Bulking agent** | **Ratio sewage sludge/bulking agent** | **Effect on the composting process** | **References** |
|---|---|---|---|
| Fly ash<br>Sawdust | 1:0.3<br>1:0.2 | Fly ash and sawdust used as bulking agents reduced 16 PAHs availability by 83–88% | Oleszczuk et al. [88] |
| Zeolite | 4:1 | Composting with clinoptilolite reduced the mobility of Zn by up to 46%, Cu by 66%, Mn by 55%, Ni by 63%, and Pb and Cd by 60–70% | Zorpas and Loizidou [89] |
| Wood chips of Acacia dealbata | 2:1<br>3:1 | Acacia wood extended thermophilic phase (65 vs. 42 days) and reduced the mobility of Zn, Cu, Cr, Ni, Pb by 38–44%. | Yañez et al. [90] |
| Green waste screening<br>Grass clippings<br>Mixture of crushed hardwood materials and ground dry leaves<br>Crushed wood pallets<br>Bark<br>Corn stalks | - | Bulking agents reduced N losses (−24 to −39%) and increased N availability (up to +11% of total N) | Doublet et al. [91] |
| Pumice and<br>Pumice decorated with sucrose | 1:1.2<br>1:1<br>1:0.8<br>1:0.6<br>1:0.4 | Use of pumice enhanced mesophilic microbial growth, promoted organic matter degradation, and reduced $NH_3$ emissions by 21%, with sucrose-modified pumice reducing nitrogen loss by 43%. | Wu et al. [92] |
| Biochar<br>Zeolite<br>Lime | - | Biochar + zeolite (12% biochar w/w): reduced $N_2O$ emissions by 95%, $CH_4$ by 93-95%, $NH_3$ by ~8-65%, enhanced organic matter humification (+35-42% humic-acid, +24-28% fulvic-acid). | Awasthi et al. [93] |
| Straw<br>Plantain leaf<br>Corncob<br>Sunflower stalk | 1:0.6 | Corncob or sunflower stalk (40%) led to 38% organic Matter degradation, 30% dry Matter loss, peak temperature of 64 °C, and pH Maintained between 6 and 7. | Uçaroğlu et al. [94] |
| Sawdust modified with rice straw biochar | - | Bulking agent (10–20%) enhanced cellulase and peroxidase activity, inhibited protease, and strengthened enzyme–microbial community relationships | Du et al. [95] |
| Wood chips Ceramsite<br>Vegetable ash | 3:1<br>4:1 | Wood chips (3:1): achieved 13 → 67 °C in 2 days, maintained > 55 °C; moisture decreased from 78–50%, volatile solids from 61–47%; ceramsite failed to initiate composting; vegetable ash reduced moisture but slowed heat release. | Lu et al. [85] |
| Tree leaves<br>Wheat straw<br>Pistachio hull wastes | - | Bulking agents (added at 10–45%) reduced mobility factor of As to 18–19%, Pb to 5–8% and Cr to 1–7%. | Saffari et al. [96] |

et al. [97], studied the effects of municipal sewage sludge composted with sawdust on a sandy Eutric Cambisol, applying it at rates of 7.5, 15.0, 22.5 and 30 g $kg^{-1}$. Incubation and pot experiments were conducted to evaluate the efficacy of composted sewage sludge (CSS) on soil N and P availability and plant nutrition. This study determined that, compared to conventional water-soluble fertilisers, CSS did not increase either soil mineral N or organic P and had minimal effects on the labile forms of P. While the efficiency of total applied P was 17% for the labile forms of soil and 4.8% for the resin extractable fraction, CSS significantly increased the inorganic P extractable from hydroxide and the non-extractable fraction of soil P. The increase in these forms of P is Mainly Due to the redistribution of the P transported by CSS between the calcium-bound and the more retained forms. While dry matter yield, N content and N uptake by the ryegrass were not improved by the CSS treatment, the P content of the plant increased at the second cut, but this was not a lasting effect, as no significant Changes were recorded in subsequent cuts. The total P uptake increased from 14.1 to 20.2 mg $pot^{-1}$, but the recovery of P by the ryegrass was minimal, with an average value of 2.5% of P transported by CSS. The study concluded that moderate application of CSS does not significantly improve crop growth but contributes to nutrient recycling without any threat to N and P loss to the environment. In another study, Debosz et al. [98] examined the effect of anaerobically digested sewage sludge and domestic compost on soil fertility through an 11-month incubation experiment and a 3-year field trial. These studies showed that most soil improvers have positive effects on soil properties, such as potentially mineralizable N and resin-extractable P. These effects were, in most cases, moderate and temporary, occurring mostly in the first weeks after application. This revealed that soil responses are more rapid under variable field conditions than under constant laboratory conditions. Overall, the study concluded that compost and sludge amendments increase soil fertility, thus indicating the need for evaluation systems to focus on the initial post-application phase for a better understanding of their impact during critical growth periods.

Vaca et al. [99] investigated the effect of sewage sludge, CSS and inorganic fertiliser applications on soil properties, maize (*Zea mays* L.) productivity and grain quality, particularly on Ni, Cu and Zn contents in soil and Maize grain. The application rate of sewage sludge and CSS was 18 Mg $ha^{-1}$, while inorganic fertiliser was applied at a proportionate dose with an N-P-K formulation of 150-75-30. In all cases, better results were observed with sewage sludge and CSS than with inorganic fertilisers. Both sewage sludge and CSS increased maize productivity without increasing heavy metal concentrations in the grain. Grain quality parameters with safe limits for human consumption included starch, protein and fibre content. Abdul Khaliq et al. [100] applied 22 kg of CSS and 0.5 kg of mineral fertiliser to monitor their effects on soil quality and growth of radish (*Raphanus sativus* L.) and bean (*Phaseolus vulgaris* L.). The application of CSS showed higher yield rates than inorganic fertilisers, mainly due to the higher availability of nutrients, especially total N. Chemical analyses revealed no accumulation of heavy metals either in the soil or in the test crops. However, due to the short duration of the experiment, the authors recommend further long-term studies (at least five years) to understand the impact of the application of CSS on soil fertility and crop yield, with positive implications for sustainable agriculture. Furthermore, the positive impact of CSS on soil quality and plant growth was related to the amount applied. In an experimental trial, Elsalam et al. [55] studied the effectiveness CSS as a fertiliser for different soils focusing on the effects on organic matter, N forms, and the growth of maize (*Zea mays* L.) and broad beans (*Vicia faba* L.). The research found that the application of CSS increased maize yield and soil organic carbon content, especially after maize harvest, which showed a residual effect. Nitrogen forms showed different trends between crops, with an increase in $NO_3$-N after maize harvest and an increase in $NH_4$-N after bean harvest. Total nitrogen for maize was higher during the growing period than after harvest, while soil nitrogen levels for beans were stable. Heavy metal concentrations in the sludge were within safe limits, making it safe to use in agriculture. Further recommended research therefore concerns the optimal application rates of sludge and its combinations with other organic fertilisers, in line with sustainable agricultural practices and the EU Action Plan for the Circular Economy.

### 5.4 Energy and materials recovery

Sewage sludge presents a dual challenge and opportunity for sustainable resource management. On one hand, its disposal poses environmental and logistical concerns, while on the other hand, it contains valuable energy and materials that can be valorised. In this framework, energy and material recovery from SS offer a promising pathway towards environmental stewardship and resource efficiency. Through advanced technologies and innovative approaches, SS can be transformed into renewable energy sources, biofuels, and high-value materials. The following sections will provide a comprehensive overview of the benefits and potential of energy and material recovery from SS, highlighting its significance in the context of sustainable waste management and circular economy principles.

#### *5.4.1 Opportunities for energy recovery from sewage sludge*

Biogas production through the anaerobic digestion process is perhaps the most widely adopted approach for energy recovery from SS. Anaerobic Digestion (AD) is a microbial mediated process by which organic matter is transformed into simple compounds in an oxygen-free environment. AD consists of four stages: (i) hydrolysis of complex organics into volatile fatty acids (VFA); (ii) acidogenesis of VFA into alcohols; (iii) acetogenesis of such compounds with the production of acetic acid, hydrogen, and $CO_2$; (iv) methanogenesis in which the acetic acid, hydrogen, and carbon dioxide are converted into biogas. Biogas consists of methane (50–70%), carbon dioxide (30–50%), and trace of minor gases such as nitrogen, hydrogen, hydrogen sulphide, and water vapor [101]. In addition to the major components, biogas derived from sewage sludge may also contain trace contaminants that pose operational and environmental challenges. Among the most relevant are volatile organic sulfur compounds, such as mercaptans (thiols), and volatile silicon compounds, known as siloxanes. Although typically present at low concentrations (ppm or ppb), these compounds can cause significant issues during biogas combustion. Therefore, biogas cleaning is essential before its use in energy systems or grid injection.

Biogas yield and quality strongly depend on the type of sludge and process operating conditions [102]. For instance, primary sludge produces a higher biogas yield than secondary sludge [103]. Overall, the methane yield is generally Limited within a range of 0.19–0.24 Nm $^3CH_4$ per kg of sludge (as dry weight) [100]. Enhancing the efficiency and energy recovery potential of the digestion process can be achieved through different preprocessing and postprocessing techniques [104]. Recent studies have focused on enhancing biogas production by improving the hydrolysis stage, which is often the limiting step in anaerobic digestion. Various pretreatments, including ozonation, sonication, wet oxidation, microwave, and thermal treatments, have been proposed and extensively investigated in the literature [105, 106]. Physical and mechanical pretreatments aim to disintegrate sludge flocs, thereby increasing the specific surface area available to methanogenic bacteria and enhancing biogas productivity. Thermal hydrolysis was among the initial methods explored for this objective. Augmenting substrate disintegration is achievable through elevated temperature (> 100 °C) and pressure (˃ 10 bar), leading to an improvement in methane production of about 90% [108]. Nonetheless, excessive temperatures should be avoided as they could harm methanogenic activity. Ultrasonication creates cavities that implode, generating hydro-mechanical shear forces that disrupt the sludge structure and potentially increase biogas production by 40–58% [109], although this method is associated with high energy costs and conflicting results. Microwave irradiation, operating within specific wavelengths, improves organic compound solubilization and can increase biogas production by 30–50%, although Managing the process can be challenging. Electro-kinetic disintegration involves applying a high-voltage electric field to disrupt cell walls, leading to a potential 30% increase in biogas production [109]. Chemical pretreatment is considered the most efficient method for enhancing methane production. Alkali pretreatment, involving reagents such as NaOH, KOH, and $Mg(OH)_2$, solubilizes organic matter by disrupting cells and extracellular polymeric substances (EPS), potentially increasing methane production by up to 80% [111]. Furthermore, pre-oxidation methods, such as ozone and peroxidation using $H_2O_2$, have shown promising results, with ozone treatment leading to a potential 200% increase in biogas production [111]. Despite numerous efforts to advance anaerobic digestion worldwide, key gaps remain for future research. More cost-effective pretreatment techniques need to be developed, and studies should expand from pilot-scale to full-scale implementations. Additionally, exploring cost-effective enzymes and bioaugmentation methods could enhance AD performance [112].

Historically, biogas has been conventionally converted into both electrical and thermal energy through combined heat and power (CHP) generators for the WWTP's own consumption. However, upgraded biogas, with the aim of increasing the methane percentage to above 96%, is of greater value since it could be stored and/or transferred through natural gas networks. Biogas upgrading is achieved by removing $CO_2$ and other impurities. Biomethane can contain as much as 51 MJ/Nm$^3$, which is much higher than the original calorific value of biogas (30–40 MJ/Nm$^3$)[104]. In Many EU countries biomethane is already injected into the natural gas network or used as automotive fuel in Line with the recent REPowerEU plan. The total cost of biogas production and upgrading ranges from € 50–120/MWh, although economies of scale could reduce the capital costs [113].

An alternative pathway to biogas and biomethane generation from sewage sludge is the production of biohydrogen. Bio-$H_2$ could be achieved by photo-fermentation (PF) and dark-fermentation (DF) processes involving the conversion of sludge organics into hydrogen, carbon dioxide and other minor compounds. Dark fermentation process is more commonly used than photo-fermentation as the latter has several process drawbacks (e.g., slow kinetics, high footprint). The difference between DF and AD lies in the inhibition of methan-producing bacteria, thus avoiding VFA and $H_2$ conversion into methane. Generally, the application of several operating strategies, such as pH adjustment, and temperature control, either individually or in combination, is used to enhance biohydrogen production. However, current hydrogen production technologies from sewage sludge are limited by production costs (5.6–11.4 \$ kg$^{-1}$) [114] and low yields (2–3 mol/mol of glucose) [115].

Besides the above opportunities, SS represents a noticeable source of chemical energy [116]. This could be recovered through thermochemical processes, after or in combination with AD processing.

Sludge incineration is a well-established method to recover heat and electricity. Incineration of SS, alone or in combination with municipal solid waste (co-incineration), produces thermal energy that is used to generate electricity but also some pollutants that require appropriate treatment before their disposal into the environment. Direct incineration of SS is reported to be about 12–16 MJ/kg [118]. To enhance energy production, efficient sludge drying processes are required [118].

Pyrolysis and gasification are additional pathways for energy recovery from SS. Pyrolysis involves organic matter decomposition in an oxygen-free environment, producing gas (syngas), liquid (bio-oil), and solid residue (char) by-products. Typically, pyrolysis is carried out on sludge after anaerobic digestion, to maximize energy recovery. Bio-oil can be used as fuel (17 MJ/kg), as well as a source of chemical products, whereas biochar (energy content up to 18 MJ/kg) can be used as a soil enhancer or adsorption material for pollutant removal [119].

Hydrothermal carbonization (HTC) of SS represents a method for reducing waste volume while offering opportunities for energy generation and valuable product extraction. Notably, HTC boasts advantages over conventional dry thermal treatments as it eliminates the need for prior drying. During HTC, the organic matter in the SS undergoes complex chemical reactions, including dehydration, decarboxylation, and polymerization, leading to the formation of a carbonaceous material known as hydrochar or biochar. This hydrochar is rich in carbon and has properties similar to those of coal or charcoal (15 MJ/Kg) [123]. The process typically operates at temperatures ranging from 180 to 250 °C and pressures between 10 and 50 bar. It is reported that the production of a biofuel through HTC can be more cost-effective than conventional thermal drying [123].

In hydrothermal liquefaction (HTL), sludge is subjected to higher temperatures (250–400 °C) and pressures (10–25 MPa) than HTC to convert organics into a mixture of water-soluble organics, gases, and bio-oil. The bio-oil produced through HTL can be further refined and upgraded to produce transportation fuels, such as biodiesel or renewable diesel, or used as a feedstock for various chemical processes [124].

### *5.4.2 Materials and value-added products from sludge*

Among several materials that could be recovered from sewage sludge, nutrients (N and P) represent one of the most consolidated solutions. During the last few years, due to economic reasons and the intensive use of fertilizer in modern agriculture, the demand for nitrogen and phosphorous has increased. Different processes, as mentioned above, can be utilized for the recovery of nitrogen and phosphorus. Indeed, they are generally extracted from liquid waste streams produced during the mechanical dewatering and thermal drying phases of anaerobically digested sewage sludge. Moreover, during thermal processes the ash produced can contain concentrated nitrogen and phosphorus, which can then be recovered and reused. Generally, struvite precipitation by adding magnesium ions in the form of $MgCl_2$ is the most adopted method to recover N and P from these streams [125]. Struvite offers slow-release fertilization benefits, representing an excellent alternative to produce inorganic fertilizers or for other industrial applications [126]. More recently, the use of membranes was successfully implemented for the recovery of ammoniacal nitrogen from sludge [127]. As an alternative, fertigation may be another possibility to directly reuse ammonium and phosphate-enriched effluent [111].

Sewage sludge has found widespread application in the production and reclamation of value-added products. Polyhydroxyalkanoates (PHAs), are biopolymers synthesized by certain bacterial strains that could offer an eco-friendly alternative to petroleum-based plastics. However, the current method entails using pure microbial cultures and pure organic substrates, although leading to high PHA accumulation yields (> 85% w/w) involves high operational expenses [128]. Utilizing sewage sludge as a mixed microbial culture has shown promising results [129]. Currently, the main issues to overcome in this process include enriching the mixed microbial culture with PHA-storing microorganisms and providing organic feedstocks enriched in volatile fatty acids [130].

In addition to PHA, other biopolymers can be extracted from sewage sludge. Bioflocculants, or extracellular polymeric substances (EPS), are naturally produced by bacteria during metabolism, facilitating their aggregation into flocs. Such bioflocculants are widely employed in wastewater treatment for heavy metal removal due to their safety, biodegradability, and minimal secondary pollution risk [131]. Among bioflocculants, alginate-like exopolysaccharides (ALE) extracted from granular sludge present a promising solution. Alginate finds application as a thickener, emulsion and foam stabilizer, encapsulation agent, gelling agent, and in film and synthetic fiber formation, among other uses [132].

Several studies have confirmed the presence of heavy metals in sewage sludge. Metals present in the wastewater can accumulates in sewage sludge during physical and biological processes. In addition, other valorisation techniques (e.g., thermo-chemical processes) can concentrate metals into biochar/ash fractions, from which individual metals could be extracted [133]. Several methods, including chemical leaching, bioleaching, electro-kinetic application, and supercritical fluid extraction, have been investigated for the removal of heavy metals from sewage sludge [134]. Typically, metal ions tend to form strong bonds with cell surfaces under neutral pH conditions. Lowering the pH can promote desorption, aiding in their recovery. Other recovery possibilities could be referred to other organic compounds, such as proteins, enzymes, biopesticides and lipids [103].

Another promising avenue for sewage sludge valorisation within the framework of the circular economy is its use as a raw material for the production of construction materials, such as bricks, lightweight aggregates, ceramics, and cement. This strategy offers a dual benefit: it reduces the amount of waste requiring disposal and substitutes virgin raw materials in energy-intensive industries like building materials manufacturing. Recent studies have shown that thermally treated or dried sewage sludge can be incorporated into clay bricks without compromising the mechanical properties of the final product. For instance, in a recent study by Chang et al. 2020 [136] it was demonstrated that substituting up to 20% of clay with dried sewage sludge led to bricks with acceptable compressive strength and improved thermal insulation, as well as lower production density and reduced environmental footprint. In addition, the same study reported that sewage sludge (≤ 15%) can replace raw material in eco-cement production with similar mechanical performance, while making it a viable alternative for reducing cement consumption and associated $CO_2$ emissions.

Another growing application involves using sludge-derived ash or pyrolysed sludge as a feedstock for lightweight aggregates (LWA). These materials have been successfully sintered to produce aggregates suitable for use in non-structural concrete or insulation layers, offering good bulk density and water absorption properties [136].

Another important application is the valorization of sewage sludge ashes as incineration products. The incineration of sewage sludge generates significant quantities of sewage sludge ash (SSA), which not only contributes to volume reduction and pathogen elimination, but also offers opportunities for recovering critical raw Materials. Among them, phosphorus is of particular interest due to its essential role in agriculture and its Limited global availability. Several studies have reported that SSA can contain up to 10% phosphorus by weight, primarily in the form of inorganic phosphates, making it a promising secondary source of this strategic element [137]. Phosphorus recovery from SSA can be achieved through various processes, including acid leaching, thermal treatment with additives (such as calcium or magnesium), and precipitation techniques like struvite crystallization. A literature study demonstrated that sewage sludge and its incineration ash can be used to produce lightweight expanded clay aggregates (LECA)-like Materials. Indeed, incorporating 5–15% dry sewage sludge as an expanding agent significantly increases total and closed porosity while reducing bulk density [138]. Based on the above, it arises that the biorefinery concept holds promise for advancing sustainable wastewater treatment by integrating resource recovery, energy production, and environmental protection objectives. Nevertheless, this concept has several limitations related to variability in wastewater compositions, high energy requirements, and the need for the integration of high-technological processes with existing wastewater treatment plants. Moreover, regulatory frameworks and market dynamics could impact the economic feasibility and attractiveness of investing in biorefinery-based solutions for SS handling.

## 6 Economics of sewage sludge management

The SS management has become a major challenge over the last few decades in relation to increasing production and the potential offered as a source of agricultural nutrients and energy [139]. The various technical options for SS treatments, to generate energy or for nutrients recovery, have been widely studied worldwide especially in relation to their environmental impact. In contrast, the research focus on the economic feasibility and cost-effectiveness SS treatments is still limited. Some of the extant studies have compared the economic feasibility of the most common used technical options in order to identify the most effective handling scenario. For example, Zhuang et al. [140] compared the life cycle cost of four SS treatments (composting, anaerobic digestion, pyrolysis, and incineration) and three disposal methods (landfilling, agricultural application, and energy recovery). The results showed that composting had the lowest life cycle cost, due to low capital investment, followed by pyrolysis, which, generated the highest income due to the sale of by-products (bio-oil and biochar), the energy recovered from biogas, and then the anaerobic digestion. In contrast, incineration was the most expensive solution due also to ash residues management and transport costs. Similarly, Arias et al. [141] comparing three different scenarios which differ in relation to the pre-treatments implemented (chemical or thermal pyrolysis), to improve biogas production, and two post-treatments (composting unit or incineration) found that thermal pyrolysis is more cost-effective compared to the conventional and chemical treatments. This is because sludge can be applied directly to agricultural soil, and for the higher biogas production which offsets the higher electricity costs compared to the other scenarios studied, resulting in a lower payback time even in the case of incineration. It should be noted that while anaerobic digestion effectively reduces the organic load and generates renewable energy in the form of biogas, it does not completely resolve the sludge disposal issue. A significant amount of digestate remains, which must be managed through additional processes such as composting, land application, or thermal treatments. Another study emphasised the cost-effectiveness of the anaerobic digestion treatment, which offers cost advantages but requires long processing times [142]. In light of this, Medina-Martos et al. [143] compared an integrated system of anaerobic digestion (AD) and hydrothermal carbonization processes with anaerobic digestion in a conventional system. Results showed that although the integrated system has a better environmental Life cycle performance, it generated 42% higher costs than the conventional AD system due to the higher investments required that are not compensated by higher revenues due to the low market value of hydrochar. These results confirm the significant influence of energy recovery, raw materials and outputs prices on the economic feasibility of different technical options. This consideration is supported by Xu et al. [144] who, comparing 13 different SS treatment scenarios in China, found that the most economically viable treatment scenario includes processes such as gravity thickening, anaerobic digestion, and dewatering, which improve energy recovery and reduce volume before final disposal or further valorisation. Other studies have focused on single treatments, such as SS incineration [41] and anaerobic digestion [145]while considering different scenarios for each of them. For example, Xiao et al. [41] studied the energy, environmental and economic impacts of SS incineration in China considering four different scenarios: mono-incineration with natural gas; co-incineration in coal-fired power plant; co-incineration in municipal solid waste (MSW) incineration plant; and co-incineration in cement plant. The results showed that the mono-incineration scenario is the most expensive solution due to both plant construction and gas costs. On the other hand, energy costs had a relatively low incidence (8%) especially compared to the co-incineration scenarios, where they varied between 38% and 46%, pointing out how the economic efficiency of the co-incineration alternatives is highly dependent on fluctuations in electricity prices. Looking at the anaerobic digestion process, Li et al. [145] compared five different processes and found that the thermophilic high-solids anaerobic digestion (THSAD) has the highest net present value (NPV) due to the higher energy and natural gas recovery compared to conventional anaerobic digestion (CAD). Ultimately, Huang et al. [146] analysed alternative sludge disposal technologies and in particular incineration, landfilling and advanced oxidation process (persulfate oxidation) for sludge degradation and nutrient recovery. The comparison showed opportunities of the persulfate oxidation which required low investment costs, a simpler process for resource recovery and the possibility of on-site treatment, which cuts transport and dewatering costs that take up a large share of landfilling and incineration processes, but at higher chemical costs.

In summary, as these studies reveal, the economic feasibility of technical solutions presents opportunities and barriers, mainly related to the dynamics of the markets for energy, gas, raw materials and the products and by-products obtained.

## 7 Sludge effects on soil properties

The composition of the wastewater determines the chemical and biological composition of the sludge. Sewage sludge, which is generally rich in organic matter and plant nutrients such as nitrogen, phosphorus, and calcium, can improve various physical, chemical, and biological properties of the soil, including porosity, aggregate stability, bulk density, and soil fertility [147] (Fig. 4). Indeed, when incorporated into the soil, sludge improves water retention capacity, promotes nutrient availability, and enhances overall soil health. However, excessive application may lead to nutrient imbalances, potential contamination, or soil compaction. Therefore, applying sludge to soil could have a significant impact on different soil parameters and be relevant in both agricultural and environmental contexts. The following paragraphs will aim to provide an exhaustive overview of the sludge application effects on the main soil properties.

### 7.1 Soil physical properties

Soil amendment is expected to reduce soil bulk density as a consequence of the lower density of organic matter compared to mineral particles, especially when initial organic matter content is low [148]. However, bulk density reduction is also owed to the improved aggregation, which leads to stable aggregates along with inter- and intra-aggregate pores [149].

A significant negative correlation between organic amendment dose and bulk density was reported, among others, by Bondì et al. [150]Głąb et al. [151]and Zuo et al. [152]. However, opposite results occurred with a bulk density of a silty loam soil that remained unaffected by 18-years application of dehydrated urban sewage sludge [153].

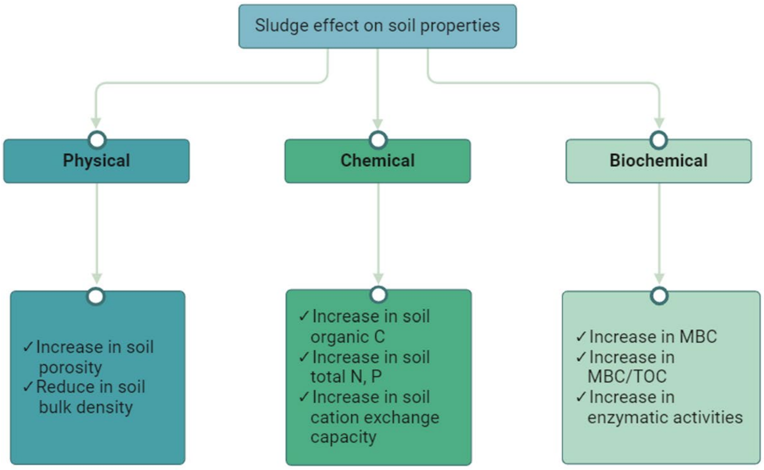

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**Fig. 4** Main effect of sewage sludge on physical, chemical and biochemical soil properties

Zuo et al. [152] evaluated the long effect of SS applied at the rates of 0, 25, 50, 125, and 250 t ha$^{-1}$ on selected soil physicochemical properties, yields and quality of sweet sorghum (*Sorghum bicolor* L.) cultivated in newly reclaimed mudflat saline-alkaline soil. Sewage sludge amendment increased the percentages of water-stable aggregates, especially at higher application rates (≥ 25 t ha$^{-1}$). Such an increase was probably due to release of carbohydrates and organic carbon during process of OM-rich SS biodegradation that facilitated formation of soil aggregates. Bulk density decreased as SS rates increased, improving soil structure and reducing the tendency for compaction. The SS, rich in organic matter and nutrients, served as a soil improver, enhancing aggregate stability, pore space and the overall physical properties of the soil.

Increased total soil porosity implies that more water is retained by soil. However, the influence of organic matter addition on the plant available water (PAW), i.e. the difference between field capacity (FC) and wilting point (WP), is not straightforward given that any increase in FC is presumably counterbalanced by that of WP [148]. As a matter of fact, an increase in PAW implies a modification of pore size distribution with increasing capillary pore volume and decreasing drainable and/or non-capillary porosities [154].

A mixture of maize straw and SS significantly affected the PAW of a loamy sandy soil (81% sand, 14% silt, and 5% clay) that increased from 0.027 to 0.043 cm$^3$/cm$^3$ when the fertilizer rate increased from 0.5 to 4% on a mass basis [148]. Compared with Maize straw feedstock, the addition of SS increased the volume of large pores and reduced the volume of pores with diameters below 50 µm thus resulting in increased PAW. Similar results were obtained by Ma et al. [155] who found that the addition of dried SS to a silty loam increased non-capillary porosity, indicating that it enhanced gas permeability of the soil with positive impact on crop growth.

In a field study aiming at comparing the effects of different organic amendments in restoring the physical and chemical properties of a degraded clay soil, Soria et al. [156] applied SS from anaerobic mesophilic digestion, dehydrated by centrifuges and Heat-dried at 70 °C at a rate of 147.4 t ha$^{-1}$. They found that SS addiction increased the soil water retention thus resulting in soil moisture content at 3 cm depth higher than the control (i.e., not amended soil) by a factor of 1.3–1.6. As a drawback, the favoured metabolic activity of soil microorganisms increased the mineralization rates potentially failing to guarantee a long-term nutrient reserve and soil fertility.

Some observations indicated that the relationship between SS addition and the improvement of physical indicators was not straightforward. For example, a long-term experiment on a calcareous clay-loam proved that PAW was not significantly different from the mineral fertilization when the common agronomic dose of 20 t ha$^{-1}$ per year of SS was applied [157]. Applying an extreme dose of 80 t ha$^{-1}$ per year caused the PAW halving. They concluded that the use of extreme doses of SS needs further examination, as it may have deleterious effects in terms of soil structure.

Differently might be the role of biochar obtained from sewage sludge. Indeed, the porous structure of biochar improves water and nutrient retention within the soil matrix. In particular, micropores contribute significantly to the adsorption surface and overall retention capacity, while mesopores play a crucial role in facilitating liquid-solid interactions [158]. In addition, biochar enhances the formation of stable aggregates promoting water infiltration, root penetration, and overall soil aeration. The incorporation of biochar obtained from SS into soil can affect its bulk density, potentially impacting water absorption and retention, root growth processes and soil fauna. This occurs because biochar from SS, as other biochar, is characterized by lower density than many soil minerals and a macro- and micropores arrangement that can capture air or water, leading to a significant decrease in particle density [66].

### 7.2 Soil chemical properties

The primary constituents of sewage sludge are nutrients, organic matter, micro- and mesofauna, and contaminants. Organic matter content has gradually decreased in Mediterranean agricultural soils in the last decades. Consequently, using these kinds of resources could be an effective strategy for addressing these wastes. Indeed, an improvement in the physicochemical characteristics of the soil and an increase in nutrients were reported by Singh et al. [159] following SS application. Sludge introduces a variety of nutrients, such as nitrogen and phosphorus, and organic Matter to the soil, with a positive impact on chemical soil properties. The organic Matter content in sludge can reach 40–50% [162]. The organic compounds in sludge acts as natural fertilizers, enhancing soil fertility and supporting plant growth. Several articles in the literature support these statements. According to Shan et al. [160]SS applied to mudflats salt-affected soil increased organic carbon (OC) content and total nitrogen and phosphorus, with increasing SS application rates. Such improvement in soil fertility, in turn, increased rice plant biomass and yield. Consuegra et al. [24] reported an increase in organic Matter content from 1.2 to 2.2% in a sandy vineyard soil. Sewage sludge not only affects organic matter and thus soil structure, but also improves, although sometimes worsens, other chemical properties such as soil reaction and electrical conductivity (EC). Indeed, there is a number of published studies describing the role of SS on pH and electrical conductivity dynamics in the soil. The mobility, adsorption, precipitation, solubility, and availability of plant nutrients and pollutants in the agricultural soils are all influenced by the pH, which has a significant impact on their fertility and quality. According to Baghina et al. [23]soil reaction increased from 7.57 units in control till 7.74 units in soil amended with 40 t ha$^{-1}$. There is a clear positive correlation between the amount of SS added and pH changes. This correlation has been explained by the content of calcium carbonate in the sludge. However, this finding disagrees with other studies which have suggested that pH decreased when SS is applied to the soil [161–163]. The decrease in soil pH can essentially be related to the low pH of the sludge and to the organic carbon that is provided by the sludge, which produces organic acids following its decomposition. Soil EC is a measure of salinity in the soil. It also influences agricultural yields, plant nutrient availability, and the metabolic processes of soil microorganisms. According to several studies, the general trend is an increase in EC following the addition of sludge. In fact, the organic acids produced as a result of the decomposition of the sludge would lead to an accumulation of soluble salts in the soil [161]. Such an increase in EC has been reported by Eid et al. [162]. Indeed, they observed an increase in EC of 93.5% in soils amended with the 30 g kg$^{-1}$ dose of sludge compared to unamended soils.

However, the effects of SS on soil chemical properties depend on several parameters, including sludge composition, application rates, and soil characteristics. Although SS has the potential to provide nutrients, excessive application can cause nutritional imbalances and heavy metal accumulation in the soil. Therefore, by carefully adjusting application rates and considering soil conditions, the beneficial effects of SS on soil quality can be maximised, while reducing potential negative implications.

### 7.3 Biochemical soil properties

Soil biochemical properties, such as microbial biomass, main microbial groups, and enzyme activities, are key indicators for assessing soil quality, functioning and health [164]. Indeed, soil microorganisms, Like fungi and bacteria, play a crucial role in more than 90% of key soil functions, such as nutrient turnover through organic matter mineralization and/or nutrient transformation and storage [26]. Biochemical indicators are often preferred over physical and chemical indicators because of their rapid response to changes in soil conditions, such as tillage, fertilization and soil amendment.

Soil biochemical response to SS application depends on several factors, including soil and SS characteristics, application rate, and climatic conditions [165]. Understanding the effect of SS application on soil biochemical properties is crucial for assessing its potential benefits and risks in agricultural and environmental contexts.

Several authors have reported the positive effect of SS application on soil biochemical properties (e.g [161, 166, 167]). (Table 5), although several studies point to possible problems related to heavy metal contamination, introduction of pathogens, and the presence of a wide spectrum of toxic and harmful micropollutants, such as pesticides, polycyclic aromatic hydrocarbons (PAHs), personal hygiene products, detergent and pharmaceutical residues, etc [70, 168–171]. Since most microbial populations are heterotrophic and react quickly to new amendments, SS amendment increases both the amount of microorganisms and their activity [158, 172]. High amounts of organic matter in SS provide a latent mass and energy source for a variety of heterotrophic microorganisms, which led to higher microbial biomass carbon (MBC) and activity values during the initial days of incubation. However, over time, the readily available carbon ran out, resulting in decreasing MBC values [173].

#### *7.3.1 Soil microbial biomass*

Numerous studies have demonstrated that SS application can significantly impact soil microbial biomass. Increases in MBC and microbial biomass nitrogen (MBN) have been reported following sludge application by Dhanker et al. [161]. They demonstrated, through a laboratory incubation test, that higher concentrations of SS result in higher levels of MBC and MBN. Specifically, the highest concentration tested (corresponding to 50 Mg ha$^{-1}$) led to more than tripled MBC values and more than doubled MBN values compared to the control soil, within just 15 days after application. Moreover, the soil with the highest application of SS after 90 days of incubation showed a microbial quotient (MBC/TOC) 27% higher than that of unamended soil. The increase in microbial quotient suggested that SS provided organic matter and nutrients that supported microbial growth [174]. Mondal et al. [166] also reported that in the 5–15 cm deep soil layer MBC increased from 436 to 838 mg kg$^{-1}$ after SS application (15 Mg ha$^{-1}$), which represents a 92% increase compared to unamended soil.

However, several authors have reported on the possible risks and negative effects associated with the use of SS as soil amendments, mainly related to the presence of organic contaminants and heavy metals. For example, Charlton et al. [175]through a meta-analysis, showed that the presence of Heavy metals in SS, such as Cu and Zn, resulted

**Table 5** Effect of sewage sludge amendments on soil microbial biomass and enzyme activities

| Parameters | Type of SS | Application rate | Location | Effect | References |
|---|---|---|---|---|---|
| Soil microbial biomass | Urban SS | 50 t ha$^{-1}$ | Lab experiment (15 days) | More than tripled MBC values and more than doubled MBN values, within just 15 days after application. | Dhanker et al. [161] |
| | Urban SS | 15 t ha$^{-1}$ | New Delhi (India) | MBC in 0–5 cm and 5–15 cm layer increase by 92% and 46%, respectively. | Mondal et al. [166] |
| | SS from citrus processing industry | 10 t ha$^{-1}$ | Pot experiment, Palermo (Italy) | MBC increase by 17%. | Lucia et al. (2023) |
| | Urban SS | 40, 80, 120 t ha$^{-1}$ yr$^{-1}$×3 years | Sandy loam soil, Nabeul (Tunisia) | MBC and MBN increase by 45 and 110%, respectively. | Hamdi et al. [26] |
| | | | Sandy soil, Nabeul (Tunisia) | MBC and MBN increase by 108 and 142%, respectively. | |
| | Urban and Industrial SS | 10 t ha$^{-1}$ yr$^{-1}$×10 years | Jaboticabal city (Brazil) | MBC decrease of about 50%. | Melo et al. 2018 |
| | Cu and Zn contaminated digested and undigested SS | 15–20 t C ha$^{-1}$ yr$^{-1}$ | 9 sites of Britain | MBC decrease from 7 to 12%. | Charlton et al. [175] |
| | Aerobic and anaerobic SS composted with rice husk | 7.5, 23, 46 t ha$^{-1}$ | Lab experiment (14 days) | Increase by increasing SS rate (5, 9 and 21%). | Sciubba et al. [186] |
| | Compost from urban SS | 9 and 12 t ha$^{-1}$ | University of Bari (Italy) | MBC increase by about 42%, on average. | Curci et al. [179] |
| | Urban SS | 4 and 8% | Lab experiment (70 days) | MBC decrease of about 76%, with a higher SS rate. | Paz-Ferreiro et al. [176] |
| Enzyme activities | Compost from urban SS | 3, 6, 9 and 12 t ha$^{-1}$ | University of Bari (Italy) | β-glucosidase and alkaline phosphatase activity increased linearly by increasing application rates of SS, FDA hydrolysis significantly increased with SS, regardless of different amounts of amendment. | Curci et al. [179] |
| | Urban SS | 60 t ha$^{-1}$×8 years | Lubelski (south-east of Poland) | Dehydrogenase, Urease, Protease activities more than doubled and Acid Phosphatase activity more than tripled after 4 years after SS application. In the following 4 years, activities decreased. | Skowrońska et al. [167] |
| | Stabilized dewatered residual sewage sludge | 10 t ha$^{-1}$ | Silesian Upland (southern Poland) | Significant reduction in the activities of alkaline and acid phosphatases, urease and nitrification potential. | Markowicz et al. [51] |
| | Compost from urban SS and garden waste | 5, 10, 50 kg per tree | Yanqing District, Beijing, China; Amorpha fruticosa Linn. Woodland | Decrease of Urease activity with 50 t tree$^{-1}$. No differences were recorded in Phosphatase and Dehydrogenase activities. | Li et al. [184] |
| | Urban SS | 5, 10, 15 t ha$^{-1}$ | New Delhi (India) | Dehydrogenase activity increase by increasing SS application rate. | Mondal et al. [166] |

**Table 5** (continued)

| Parameters | Type of SS | Application rate | Location | Effect | References |
|---|---|---|---|---|---|
| | Urban SS | 40 and 120 t $ha^{-1}$ | Lab experiment | Protease activity and soil respiration increase by increasing SS application rate. | Hechmi et al. [187] |
| | Urban SS | 4 and 8% | Lab experiment (70 days) | β-glucosidase and Phosphomonoesteraseactivities decrease follow the SS application. | Paz-Ferreiro et al. [176] |
| | Urban and Industrial SS | 4–65 t $ha^{-1}$ | Jaguariúna, state of São Paulo, Brazil | Amylase and urease activities increase with increasing SS doses, after 4 years. | Pavan Fernandes et al. 2005 |
| | Urban SS | 5–50 t $ha^{-1}$ | Lab experiment | Dehydrogenase, alkaline phosphatase and urease activities increase by increasing SS rate, During the first 30 days of incubation. Therefore, activities decreased even if they were greater or equal to unamended soil. | Dhanker et al. [161] |
| | Urban SS | 40, 80, 120 t $ha^{-1}$ $yr^{-1}$ × 3 years | Sandy and sandy loam soil Nabeul (Tunisia) | Dehydrogenase, protease and phosphatase activities increase by increasing SS application rate after three consecutive annual amendments. | Hamdi et al. [26] |

in a decrease in MBC after 4 years of application, ranging from 7 to 12% compared to unamended soil. Also, Paz-Ferreiro et al. [176]have reported a decrease of MBC, of up to 76%, by increasing SS rate application. Therefore, sludge treatment, such as composting, before land application is often necessary. In fact, the composting process, in addition to stabilizing organic matter, sanitizes from pathogens, dissipates and degrades some organic contaminants, and decreases the mobility and bioavailability of heavy metals [177, 178]. However, generally, the application of compost from SS in fact has been shown to have a positive effect on soil biochemical properties. For example, research conducted by Curci et al. [179] revealed that compost obtained from urban sewage sludge increased MBC by about 42% with 12 Mg $ha^{-1}$ of compost application, indicating its positive impact on soil microbial biomass.

#### *7.3.2 Enzyme activities*

Soil enzymatic activities are sensitive indicators of soil biological activity and offer insights about the turnover of organic matter and biogeochemical cycles [51, 180]. While some enzymes experience increased activity due to the addition of organic matter, nutrients and exogenous microorganisms present in sludge [163, 165]others may be inhibited by factors such as heavy metals or pathogens introduced through sludge [181, 182].

For example, Curci et al. [179] reported a linear increase of β-glucosidase and alkaline phosphatase activity by increasing rates of SS application; on the other hand, although fluorescein diacetate (FDA) hydrolysis increased significantly with SS application, no differences were recorded between different application rates. Also, Hamdi et al. [26] report, after three consecutive annual amendments with urban SS, in two soils with different texture (sandy and sandy loam), showed an increase not only in MBC and MBN but also in enzymatic activities. In particular, dehydrogenase, protease and phosphatase activities increased by increasing SS application rates (40, 80, 120 t $ha^{-1}$ $yr^{-1}$), with a higher value in sandy-loam soil. Kizilkaya and Bayrakli [183]through an incubation experiment, investigated the effects of adding SS at different doses (0, 100, 200, and 300 t ha$^{-1}$) and C/N ratios (3:1, 6:1, and 9:1) on the activities of β-glucosidase, alkaline phosphatase, arylsulphatase, and urease in a clay loam soil. The maximum β-glucosidase activity was found in soils with the highest C/N ratio and sludge dose, whereas the highest activities of urease, alkaline phosphatase and arylsulphatase were observed in soil amended with lower C/N and the highest dose of sludge. When compared to unamended soil, alkaline phosphatase and aryl sulphatase activities increased over the first 30 days of incubation, before sharply declining afterwards. On the other hand, within 15 days, urease activity increased and then decreased. Similarly, Dhanker et al. [161] observed that, with increasing applications of SS, the soil experienced greater increases in dehydrogenase, alkaline phosphatase, and urease activities During the first 30 days of incubation. Thereafter, all activities decreased, although they maintained values greater than or equal to the unamended control soil. Nevertheless, some authors [51, 184, 185]have found that SS negatively alters microbial communities and, consequently, soil functionality and enzymatic activities, due to the negative effect of the presence of heavy metals and other toxic organic compounds.

#### *7.3.3 Soil microbial community structure*

Sewage sludge land application as soil amendment produces different responses on soil microbial community structure. The use of organic amendments affects the microbial development and activity as well as microbial community composition [188]. Changes in microbial community structure are widely influenced by SS quality, such as organic matter content and contaminant levels, and soil reaction. These factors play a crucial role in determining the amount and nature of microbial community shifts in the soil environment [189]. The application of SS to soil increases TOC content and its different fractions, and also affects the microbial growth and activity. Indeed, depending on the characteristics of the organic soil amendment, the soil microbiota can interact differently with the C provided by this amendment. This interaction can lead to the preferential growth of microbial groups which are well suited to the specific characteristics of the applied organic material, thus altering the composition and functionality of the soil microbial community [190]. However, the presence of heavy metals or pollutants in sludge can perturb the community balance and reduce microbial diversity [191]. Long-term applications of sludge can lead to changes in microbial populations, promoting specific groups adapted to nutrient-rich or metal-stressed conditions [192]. Moreover, soil pH is also considered to be one of the most significant determinants of microbial community structure, influencing microbial composition and functionality.

Microbial phospholipid fatty acid (PLFA) is an indicator of soil quality changes as it is sensitive to environmental changes. This technique is widely employed to assess shifts in soil microbial biomass and alterations in microbial community structure [193]. Hua et al. [194] compared the effects of long-term applications of biosolids and mineral fertilizer on soil microbial community structure using PLFA analysis in an agricultural field. The biosolids application altered the structure of soil microbial community structure not only in quantity but also in composition. The total amount of PLFA decreased by 18% in comparison with mineral fertilizer. You et al. [195] evaluated the impact of different sewage sludge biochar rates (20, 40, 60 t ha$^{-1}$) on soil characteristics and peanuts growth in a loamy sand soil. They observed an increase in total PLFA following the application of biochar from sewage sludge (SS), with no significant differences between application rates. Biochar altered the microbial community structure, mainly increasing the ratios of F/B and G+/G-. These changes are probably due to the effects on soil pH, which decreased, thus stimulating soil microbial biomass and altering the composition of the microbial community. This reduction can be attributed to the toxic elements found in biochar, which are known to suppress the composition of the microbial community. These results suggest that the application of high quantities of biochar derived from sewage sludge (60 t $ha^{-1}$) could inhibit soil microbial biomass for at least two years in sandy and loamy soils.

## 8 Concluding remarks and future perspectives

This literature review explores the opportunities involved in the application of sewage sludge, as well as the obstacles associated with its management. Excessive production of sewage sludge poses a serious concern for wastewater treatment plants due to environmental and socioeconomic factors. Several studies have demonstrated that sewage sludge represents a reservoir of energy and precious compounds. According to the latest energy and resource recovery technologies and techniques, a high recovery yield is the first step towards a sustainable alternative, on the economic, environmental and social point of view, and in a perspective of circularity of business models. The recovery of organic matter and nutrients from sewage sludge for reuse in agriculture, after specific treatment, is not only in line with the principles of the circular economy, but also a promising solution for value creation. However, up to now not all European countries have developed a proper manner to consciously recycle and reuse this source. The various processes involved in recovery also pose challenges due to their economic feasibility, sometimes entailing costs that may be unaffordable for small or medium-sized companies. In light of this, and despite the various studies examined, it has become noticeable that research into the economic feasibility and cost-effectiveness of sewage sludge remains limited, highlighting the need to conduct economic feasibility studies, also comparing different solutions, which can provide useful information to support farmers' decisions. The energy, gas, raw material, and by-products markets' dynamics could be the determining factor in this limitation. In addition to economic barriers, there are also legislative limitations that needs to be addressed. From a legislative perspective, there is a need to revise and harmonise the European directive (86/278/EEC) on the agricultural use of sewage sludge. The directive, although foundational, does not adequately incorporate recent scientific insights and has been differently transposed by Member States, leading to heterogeneous standards regarding heavy metals, pathogens, and organic micropollutants in both sludge and receiving soils. Sewage sludge and biosolids must be applied with proper guidelines and regulatory supervision to prevent contamination of agricultural soils, which is necessary for food production.

As the literature review reveals, the use of sewage sludge as a fertilizer in agriculture has several advantages in terms of soil properties and productivity. This resource is significant due to its ability to enhance soil structure, microbial activity, and fertility. However, to optimise the benefits and minimise the potential negative effects, cautious handling and strict observance with safety regulations are essential. Constant innovation and research on treatment technologies will improve the safe and efficient application of sewage sludge in agriculture, supporting environmentally safe and sustainable agricultural practices. By recycling sludge nutrients, producers can save money and reduce environmental impact by replacing or minimizing the use of mineral fertilizers in cultivation systems. Nevertheless, before using it, a cost-benefit analysis of its land application should be carried out due to the importance of labour and transport costs. However, sewage sludge effects on soil are determined by several factors, including sludge composition, application rates, and soil characteristics. While sewage sludge can offer nutrients, excessive application can lead to nutritional imbalances and the accumulation of heavy metals in soil. It has been demonstrated by different studies that transforming sewage sludge into biochar reduces the bioavailability of heavy metals and toxic sludge components. The transformation of sewage sludge into biochar is an effective method of reusing this organic waste, as it has a less impact on soil and plants.

Finally, it should be noted that sewage sludge management is part of a general framework characterised by a rapid population growth and the presence of different industries, which certainly have an impact on the generation of sewage sludge. Controlling or reducing the amount of sewage sludge produced during the wastewater treatment process should be also a goal of future research. Nevertheless, future experiments should be guided by a sustainable approach that combines technical, environmental, health, legal, economic, and social aspects.

**Abbreviations**

| | |
|---|---|
| SS | Sewage sludge |
| SSA | Sewage sludge ash |
| WWTP | Wastewater treatment plant |
| TS | Total solids |
| PE | Population Equivalent |
| MPs | Microplastics |
| PPCPs | Pharmaceuticals and personal care products |
| PFAS | Perfluorinated alkylated substances |
| PFCs | Functional product categories |
| CMCs | Fertiliser material categories |
| CEC | Cation exchange capacity |
| FA | Fulvic acids |
| HA | Humic acids |
| AD | Anaerobic digestion |
| VFA | Volatile fatty acids |
| CHP | Combined heat and power |
| PF | Photo-fermentation |
| DF | Dark-fermentation |
| HTC | Hydrothermal carbonization |
| HTL | Hydrothermal liquefaction |
| PHA | polyhydroxyalkanoate |
| EPS | Extracellular polymeric substances |
| ALE | Alginate-like exopolysaccharides |
| MSW | Municipal solid waste |
| PAW | Plant available water |
| FC | Field capacity |
| WP | Wilting point |
| LECA | Light expanded clay aggregate |

**Author contributions**

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## Consent to publish

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## References

1. Gherghel A, et al. A review on wastewater sludge valorisation and its challenges in the context of circular economy. J Clean Prod. 2019;228:244–63.
2. Khalid S, et al. A review of environmental contamination and health risk assessment of wastewater use for crop irrigation with a focus on low and high-income countries. Int J Environ Res Public Health. 2018;15:895. https://doi.org/10.3390/ijerph15050895.
3. Jones ER, van Vliet MTH. Country-level and gridded estimates of wastewater production, collection, treatment and reuse. Earth Syst Sci Data 13, 237–54. https://doi.org/10.5194/essd-13-237-2021(2021).
4. González I, et al. Assessment of the treatment, production and characteristics of WWTP sludge in Andalusia by multivariate analysis. Process Saf Environ Prot. 2017;109:609–20.
5. FAO. Wastewater treatment and use in agriculture. Irrigation and Drainage Paper 47, FAO, Rome. (1992).
6. Emmanuil C, et al. Terrestrial bioassays for assessing the biochemical and toxicological impact of biosolids application derived from wastewater treatment plants. Sci Total Environ. 2024;931:172718. https://doi.org/10.1016/j.scitotenv.2024.172718.
7. Soobhany N. Insight into the recovery of nutrients from organic solid waste through biochemical conversion processes for fertilizer production: A review. J Clean Prod. 2019;241:118413.
8. Aleisa E, et al. Recirculating treated sewage sludge for agricultural use: life cycle assessment for a circular economy. Waste Manag. 2021;135:79–89.
9. Laudicina VA, Badalucco L, Palazzolo E. Effects of compost input and tillage intensity on soil microbial biomass and activity under mediterranean conditions. Biol Fert Soils. 2011;47:63–70.
10. Collivignarelli MC, et al. Legislation for the reuse of biosolids on agricultural land in europe: overview. Sustainability. 2019;11:1–22.
11. Muscarella SM, et al. Wastewater treatment sludge composting. Curr Dev Biotechnol Bioeng Ch. 2023;6:115–36.
12. International Fertilizer Association (IFA). Fertilizer Outlook 2020–2024, Market Intelligence and Agriculture Services. (2020).
13. Sharma LD et al. Sustainable Management and Utilization of Sewage Sludge. https://doi.org/10.1007/978-3-030-85226-9_10 (2022).
14. Ammar E, et al. Environmental, economic, and ethical assessment of the treated wastewater and sewage sludge valorization in agriculture. *Interaction and fate of pharmaceuticals in Soil-Crop systems: the impact of reclaimed wastewater*. Hdb Env Chem. 2020. https://doi.org/10.1007/698_2020_606.
15. Ferrentino R, et al. Full-scale sewage sludge reduction technologies: A review with a focus on energy consumption. Water. 2023;15:615. https://doi.org/10.3390/w15040615.
16. Zhang L, et al. Overview of current biological and thermo-chemical treatment technologies for sustainable sludge management. Waste Manag Res. 2014;32:586–600.
17. Bhadrecha P, et al. Biological and thermo-chemical treatment technologies for sustainable sludge management. Sustainable Manage Utilization Sew Sludge. 2022. https://doi.org/10.1007/978-3-030-85226-9_5.
18. Giwa AS, et al. Advances in sewage sludge application and treatment: process integration of plasma pyrolysis and anaerobic digestion with the resource recovery. Heliyon. 2023;9:e19765.
19. Kominko H, et al. Sewage sludge: A review of its risks and circular Raw material potential. J Water Process Eng. 2024;63:105522. https://doi.org/10.1016/j.jwpe.2024.105522.
20. Foladori P, Andreottola G, Ziglio G. Sludge reduction technologies in wastewater treatment plants. IWA Publishing; 2010. https://doi.org/10.2166/9781780401706.
21. Collivignarelli MC, et al. Minimization of municipal sewage sludge by means of a thermophilic membrane bioreactor with intermittent aeration. J Clean Prod. 2017;143:369–76. https://doi.org/10.1016/j.jclepro.2016.12.101.
22. Iticescu C et al. The characteristics of sewage sludge used on agricultural lands. AIP Conf. Proc. 2022 (2018). https://doi.org/10.1063/1.5060681
23. Baghina N, et al. Sewage sludge fertilisation influence on main soil chemical features. J Environ Prot Ecol. 2014;15(1):217–22.
24. Bravo-Martín-Consuegra S, et al. Effect of the addition of sewage sludge as a fertilizer on a sandy vineyard soil. J Soils Sediments. 2016;16:1360–5.
25. Alvarenga P, et al. Sewage sludge, compost and other representative organic wastes as agricultural soil amendments: benefits versus limiting factors. Waste Manag. 2015;40:44–52. https://doi.org/10.1016/j.wasman.2015.01.027.
26. Hamdi H, et al. Repetitive land application of urban sewage sludge: effect of amendment rates and soil texture on fertility and degradation parameters. CATENA. 2019;172:11–20.
27. Ahmad A, Chowdhary P, Khan N, Chaurasia D, Varjani S, Pandey A, Chaturvedi P. Effect of sewage sludge Biochar on the soil nutrient, microbial abundance, and plant biomass: A sustainable approach towards mitigation of solid waste. Chemosphere. 2022;287(Part 1):132112. https://doi.org/10.1016/j.chemosphere.2021.132112.

28. Achkir A, et al. Implication of sewage sludge increased application rates on soil fertility and heavy metals contamination risk. Emerg Contam. 2023;9:100200. https://doi.org/10.1016/j.emcon.2022.100200.
29. Hassan F, et al. Microplastic contamination in sewage sludge: abundance, characteristics, and impacts on the environment and human health. Environ Technol Innov. 2023;31:103176. https://doi.org/10.1016/j.eti.2023.103176.
30. Beduk F, et al. Persistent organic pollutants in sewage sludge: occurrence, Temporal concentration variation and risk assessment for sewage sludge amended soils. KSCE J Civ Eng. 2023;27:3694–704. https://doi.org/10.1007/s12205-023-2385-x.
31. Feng J, Burke I, Chen X, Stewart D. Assessing metal contamination and speciation in sewage sludge: implications for soil application and environmental risk. Reviews Environ Sci Bio/Technology. 2023;22. https://doi.org/10.1007/s11157-023-09675-y.
32. European Commission. Public consultation on the Sewage Sludge Directive review initiative, European Commission. 2020. ; 2020. pp. 1–7. https://ec.europa.eu/environment/pdf/waste/sludge/Stakeholder%20Consultation%20Strategy%20-%20Evaluation%20of%20SSD.pdf (2020).
33. Council of the European Communities. Council Directive 86/278/EEC of 12 June 1986 on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. Official Journal L 1986;181:6–12. Council of the European Communities. Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment. *Official Journal* L 1991;135:40–52.
34. Collivignarelli MC, Abbà A, Frattarola A, Carnevale Miino M, Padovani S, Katsoyiannis I, Torretta V. Legislation for the reuse of biosolids on agricultural land in europe: overview. Sustainability. 2019;11(21):6015. https://doi.org/10.3390/su11216015.
35. The European Parliament and the Council of the European Union. Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 laying down rules on the making available on the market of EU fertilising products and amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and repealing Re. Official J. Eur. Union L 2019;170/1 (2019).
36. Kacprzak M, et al. Sewage sludge disposal strategies for sustainable development. Environ Res. 2017;156:39–46. https://doi.org/10.1016/j.envres.2017.03.010.
37. Arroyo P, Molinos-Senante M. Selecting appropriate wastewater treatment technologies using a choosing-by-advantages approach. Sci Total Environ. 2018;625:819–27. https://doi.org/10.1016/j.scitotenv.2017.12.331.
38. Marlow DR, et al. Towards sustainable urban water management: A critical reassessment. Water Res. 2013;47:7150–61. https://doi.org/10.1016/j.watres.2013.07.046.
39. Eurostat. 2022. Sewage Sludge Production and Disposal, 2.18.21 [WWW Document]. URL: https://ec.europa.eu/eurostat/databrowser/view/ten00030/default/line?la ng = en
40. Yakamercan E, Ari A, Aygün A. Land application of municipal sewage sludge: human health risk assessment of heavy metals. J Clean Prod. 2021;319:128568. https://doi.org/10.1016/j.jclepro.2021.128568.
41. Xiao H, et al. Environmental, energy, and economic impact assessment of sludge management alternatives based on incineration. J Environ Manage. 2022;321:115848.
42. Chojnacka K, et al. Management of biological sewage sludge: fertilizer nitrogen recovery as the solution to fertilizer crisis. J Environ Manage. 2023;326:116602. https://doi.org/10.1016/j.jenvman.2022.116602.
43. Kominko H, et al. The possibility of organo-mineral fertilizer production from sewage sludge. Waste Biomass Valor. 2017;8:1781–91.
44. Namdari M, Soleimani M, Mirghaffari N, et al. Effect of biological sewage sludge and its derived Biochar on accumulation of potentially toxic elements by corn (*Zea Mays* L). Sci Rep. 2024;14:5985. https://doi.org/10.1038/s41598-024-56652-8.
45. Lamastra L, Suciu NA, Trevisan M. Sewage sludge for sustainable agriculture: contaminants' contents and potential use as fertilizer. Chem Biol Technol Agric. 2018;5:10.
46. Goss MJ, Tubulleh A, Gorahoo D. A review of the use of organic amendments and the risk to human health. Adv Agron. 2013;120:275–379.
47. Cucina M, et al. Benefits and risks of long-term recycling of pharmaceutical sewage sludge on agricultural soil. Sci Total Environ. 2019;695:133762. https://doi.org/10.1016/j.scitotenv.2019.133762.
48. a Silva Souza T, et al. Toxic potential of sewage sludge: histopathological effects on soil and aquatic bioindicators. Ecol Indic. 2020;111:105980. https://doi.org/10.1016/j.ecolind.2019.105980.
49. Buckwell A, Nadeu E. Nutrient recovery and reuse (NRR) in European agriculture. A review of the issues, opportunities and actions. Brussels: Commission of European Communities; 2016.
50. FAO. World fertilizer trends and outlook to 2018. Rome: FAO; 2015.
51. Markowicz A, et al. Land application of sewage sludge: response of soil microbial communities and potential spread of antibiotic resistance. Environ Pollut. 2021;271:116317.
52. Abdoli M. Sustainable use of sewage sludge in soil fertility and crop production. In: Sustainable Manage Utilization Sew Sludge, 297–333 (2022).
53. Jamil M, Qasim M, Uman M. Utilization of sewage sludge as organic fertiliser in sustainable agriculture. J Appl Sci. 2006;6:531–3.
54. Boudjabi S, Chenchouni H. On the sustainability of land applications of sewage sludge: how to apply the sewage biosolid in order to improve soil fertility and increase crop yield? Chemosphere. 2021;282:131122. https://doi.org/10.1016/j.chemosphere.2021.131122.
55. Elsalam HEA, et al. Effect of sewage sludge compost usage on corn and Faba bean growth, carbon and nitrogen forms in plants and soil. Agronomy. 2021;11(4):628.
56. Gopinath A, et al. Conversion of sewage sludge into biochar: A potential resource in water and wastewater treatment. Environ Res. 2021;194:110656. https://doi.org/10.1016/j.envres.2020.110656.
57. Oni BA, Oziegbe O, Olawole OO. Significance of Biochar application to the environment and economy. Ann Agric Sci. 2019;64(2):222–36. https://doi.org/10.1016/j.aoas.2019.12.006.
58. Calvelo Pereira R, et al. Assessment of the influence of Biochar on rumen and silage fermentation: A laboratory-scale experiment. Anim Feed Sci Technol. 2014;196:22–31. https://doi.org/10.1016/j.anifeedsci.2014.06.019.
59. Pelagalli V, et al. Pyrolysis of municipal sewage sludge: challenges, opportunities, and new valorization routes for biochar, bio-oil, and pyrolysis gas. Environ Sci : Water Res Technol. 2024;10(10):2282–312. https://doi.org/10.1039/D4EW00278D.
60. Chen W, et al. Past, present, and future of Biochar. Biochar. 2019;1:75–87. https://doi.org/10.1007/s42773-019-00008-3.

61. Agbede TM, Oyewumi A. Benefits of biochar, poultry manure, and biochar–poultry manure for improvement of soil properties and sweet potato productivity in degraded tropical agricultural soils. Resour Environ Sustain. 2022;7:100051. https://doi.org/10.1016/j.resenv.2022.100051.
62. Fuke P, et al. Role of microbial diversity to influence the growth and environmental remediation capacity of bamboo: A review. Ind Crops Prod. 2021;167:113567. https://doi.org/10.1016/j.indcrop.2021.113567.
63. He M, et al. A critical review on performance indicators for evaluating soil biota and soil health of biochar-amended soils. J Hazard Mater. 2021;414:125378. https://doi.org/10.1016/j.jhazmat.2021.125378.
64. Khan N, et al. Biochar and environmental sustainability: emerging trends and techno-economic perspectives. Bioresour Technol. 2021;332:125102. https://doi.org/10.1016/j.biortech.2021.125102.
65. Velli P, Manolikaki I, Diamadopoulos E. Effect of Biochar produced from sewage sludge on tomato (Solanum lycopersicum L.) growth, soil chemical properties and heavy metal concentrations. J Environ Manage. 2021;297:113325. https://doi.org/10.1016/j.jenvman.2021.113325.
66. Goldan E, et al. Evaluation of the use of sewage sludge Biochar as a soil Amendment—A. Rev Sustainability. 2022;14:5309. https://doi.org/10.3390/su14095309.
67. Van Wesenbeeck S, et al. Sewage sludge carbonization for Biochar applications: fate of heavy metals. Energy Fuels. 2014;28:5318–26. https://doi.org/10.1021/ef500875.
68. Hossain MK, et al. Agronomic properties of wastewater sludge Biochar and bioavailability of metals in production of Cherry tomato (Lycopersicon esculentum). Chemosphere. 2010;78(9):1167–71.
69. Rorat A, et al. Sanitary and environmental aspects of sewage sludge management. Industrial and municipal sludge. Butterworth-Heinemann; 2019. pp. 155–80.
70. Fijalkowski K, et al. The presence of contaminations in sewage sludge–The current situation. J Environ Manag. 2017;203:1126–36.
71. Oleszczuk P. Application of three methods used for the evaluation of polycyclic aromatic hydrocarbons (PAHs) bioaccessibility for sewage sludge composting. Bioresour Technol. 2009;100(1):413–20.
72. Collivignarelli MC et al. Recovery of sewage sludge on agricultural land in lombardy: current issues and regulatory scenarios. Environ Eng Manag J 14(7) (2015).
73. Walling E, Vaneeckhaute C. Greenhouse gas emissions from inorganic and organic fertilizer production and use: A review of emission factors and their variability. J Environ Manage. 2020;276:111211.
74. Chinakwe EC, et al. Effect of temperature changes on the bacterial and fungal succession patterns during composting of some organic wastes in greenhouse. J Adv Microbiol. 2019;15(1):1–10.
75. Ayed F, et al. Assessment of physico-chemical, microbial and phytotoxic changes of various organic wastes during their composting process. J Environ Agric Stud. 2021;2(2):21–35.
76. Raza S, Ahmad J. Composting process: A review. Int J Biol Res. 2016;4(2):102–4.
77. Bhave PP, Kulkarni BN. Effect of active and passive aeration on composting of household biodegradable wastes: a decentralized approach. Int J Recycl Org Waste Agric. 2019;8:335–44.
78. Argun YA, et al. Composting as a waste management method. J Int Environ Appl Sci. 2017;12(3):244–55.
79. Cooperband LR. Composting: Art and science of organic waste conversion to a valuable soil resource. Lab Med. 2000;31(5):283–90.
80. Chen L, et al. The composting process. Dairy Manure Compost Prod Use Ida. 2011;2:513–32.
81. Meena AL, et al. Composting: phases and factors responsible for efficient and improved composting. Agric Food: e-Newsletter. 2021;1:85–90.
82. Azim K, et al. Composting parameters and compost quality: a literature review. Org Agric. 2018;8:141–58.
83. Wong JW, et al. Dewatering of saline sewage sludge using iron-oxidizing bacteria: effect of substrate concentration. Bioresour Technol. 2016;213:31–8.
84. Wu B, Dai X, Chai X. Critical review on dewatering of sewage sludge: influential mechanism, conditioning technologies and implications to sludge re-utilizations. Water Res. 2020;180:115912.
85. Lu W, et al. Performance evaluation in composting of sewage sludge with different bulking agents. J Environ Eng. 2020;146(6):05020002.
86. Tarpani RRZ, et al. Life cycle environmental impacts of sewage sludge treatment methods for resource recovery considering ecotoxicity of heavy metals and pharmaceutical and personal care products. J Environ Manage. 2020;260:109643.
87. Awasthi MK, et al. Heterogeneity of Biochar amendment to improve the carbon and nitrogen sequestration through reducing greenhouse gases emissions during sewage sludge composting. Bioresour Technol. 2017;224:428–38.
88. Oleszczuk P. Influence of different bulking agents on the disappearance of polycyclic aromatic hydrocarbons (PAHs) during sewage sludge composting. Water Air Soil Pollut. 2006;175:15–32.
89. Zorpas AA, Loizidou M. Sawdust and natural zeolite as a bulking agent for improving quality of a composting product from anaerobically stabilized sewage sludge. Bioresour Technol. 2008;99:7545–52.
90. Yañez R, Alonso JL, Díaz MJ. Influence of bulking agent on sewage sludge composting process. Bioresour Technol. 2009;100:5827–33.
91. Doublet J, et al. Influence of bulking agents on organic matter evolution during sewage sludge composting; consequences on compost organic matter stability and N availability. Bioresour Technol. 2011;102(2):1298–307.
92. Wu C, et al. Usage of pumice as bulking agent in sewage sludge composting. Bioresour Technol. 2015;190:516–21.
93. Awasthi MK, et al. Role of Biochar amendment in mitigation of nitrogen loss and greenhouse gas emission during sewage sludge composting. Bioresour Technol. 2016;219:270–80.
94. Uçaroğlu S, Alkan U. Composting of wastewater treatment sludge with different bulking agents. J Air Waste Manage Assoc. 2016;66:288–95.
95. Du J, et al. Effects of Biochar on the microbial activity and community structure during sewage sludge composting. Bioresour Technol. 2019;272:171–9.
96. Saffari M, et al. Assessment of content and chemical forms of arsenic, copper, lead, and chromium in sewage sludge compost as affected by various bulking agents. Main Group Metal Chem. 2020;43(1):56–66.
97. Coutinho J, Arrobas, Rodrigues M. Effect of composted sewage sludge amendment on soil nitrogen and phosphorus availability. Commun Soil Sci Plant Anal. 1997;28(19–20):1845–57.

98. Debosz K, et al. Evaluating effects of sewage sludge and household compost on soil physical, chemical and Microbiological properties. Appl Soil Ecol. 2002;19(3):237–48.
99. Vaca R, et al. Effects of sewage sludge and sewage sludge compost amendment on soil properties and *Zea Mays* L. plants (heavy metals, quality and productivity). Rev Intern Contam Ambient. 2011;27:304–11.
100. Abdul Khaliq SJ, et al. The effect of municipal sewage sludge on the quality of soil and crops. Int J Recycl Org Waste Agric. 2017;6:289–99.
101. Di Capua F, et al. High-solid anaerobic digestion of sewage sludge: challenges and opportunities. Appl Energy. 2020;278:115608. https://doi.org/10.1016/j.apenergy.2020.115608.
102. Werkneh AA, Gebru SB. Development of ecological sanitation approaches for integrated recovery of biogas, nutrients, and clean water from domestic wastewater. Resour Environ Sustain. 2023;11:100095. https://doi.org/10.1016/j.resenv.2022.100095.
103. Capodaglio AG, Callegari A. Energy and resources recovery from excess sewage sludge: A holistic analysis of opportunities and strategies. Resour Conserv Recycl Adv. 2023;19:200184. https://doi.org/10.1016/j.rcradv.2023.200184.
104. Oladejo J, et al. A review of sludge-to-energy recovery methods. Energies. 2019;12:1–38. https://doi.org/10.3390/en12010060.
105. Fang W, et al. Effect of alkaline addition on anaerobic sludge digestion with combined pretreatment of alkaline and high pressure homogenization. Bioresour Technol. 2014;168:167–72. https://doi.org/10.1016/j.biortech.2014.03.050.
106. Zhen G, et al. Overview of pretreatment strategies for enhancing sewage sludge disintegration and subsequent anaerobic digestion: current advances, full-scale application and future perspectives. Renew Sustain Energy Rev. 2017;69:559–77. https://doi.org/10.1016/j.rser.2016.11.187.
107. Liang J, et al. Thermal hydrolysis of wastewater sludge followed by fungal fermentation for organic recovery and hyphae fiber production. Eng. 2021;7:203–11. https://doi.org/10.1016/j.eng.2020.09.002.
108. Pilli S, et al. Ultrasonic pretreatment of sludge: A review. Ultrason Sonochem. 2011;18:1–18. https://doi.org/10.1016/j.ultsonch.2010.02.014.
109. Lee IS, Rittmann BE. Effect of low solids retention time and focused pulsed pre-treatment on anaerobic digestion of waste activated sludge. Bioresour Technol. 2011;102:2542–8. https://doi.org/10.1016/j.biortech.2010.11.082.
110. Mu W, Dagnew M. Enhancing biomethane production and phosphorus recovery from CEPT sludge through a low temperature thermal alkali and ozonation pretreatment processes. Case Stud Chem Environ Eng. 2022;5:100178. https://doi.org/10.1016/j.cscee.2021.100178.
111. Mainardis M, et al. Life cycle assessment of sewage sludge pretreatment for biogas production: from laboratory tests to full-scale applicability. J Clean Prod. 2021;322:129056. https://doi.org/10.1016/j.jclepro.2021.129056.
112. Nayeri D, et al. A comprehensive review on the recent development of anaerobic sludge digestions: performance, mechanism, operational factors, and future challenges. Results Eng. 2024;22:102292. https://doi.org/10.1016/j.rineng.2024.102292.
113. Prussi M, et al. Biomethane as alternative fuel for the EU road sector: analysis of existing and planned infrastructure. Energy Strat Rev. 2021;33:100612. https://doi.org/10.1016/j.esr.2020.100612.
114. Alam M, Fatema Nayan N. Techno-Economic Assessment of Biohydrogen Production from Dark Fermentation of Wastewater Sludge (2024).
115. Wu SY, Lin CN, Chang JS. Hydrogen production with immobilized sewage sludge in three-phase fluidized-bed bioreactors. Biotechnol Prog. 2003;19:828–32.
116. Hu M, et al. A review on turning sewage sludge to value-added energy and materials via thermochemical conversion towards carbon neutrality. J Clean Prod. 2022;379:134657.
117. Jadlovec M, et al. Pollutants production, energy recovery and environmental impact of sewage sludge co-incineration with biomass pellets. Environ Technol Innov. 2023;32:103400. https://doi.org/10.1016/j.eti.2023.103400.
118. Sorrenti A, et al. Enhanced sewage sludge drying with a modified solar greenhouse. Clean Technol. 2022;4:407–19. https://doi.org/10.3390/cleantechnol4020025.
119. Callegari A, Capodaglio AG. Properties and beneficial uses of (bio)chars, with special attention to products from sewage sludge pyrolysis. Resources. 2018;7. https://doi.org/10.3390/resources7010020.
120. Werle S. Sewage sludge gasification: theoretical and experimental investigation. Environ Prot Eng. 2013;39:25–32. https://doi.org/10.5277/EPE130203.
121. Tyagi VK, Lo SL. Energy and resource recovery from sludge: Full-scale experiences. Environ Mater Waste Resour Recover Pollut Prev. 2016;221–44. https://doi.org/10.1016/B978-0-12-803837-6.00010-X.
122. Gong L, et al. Impact of hydrothermal pre-treatment on the anaerobic digestion of different solid-liquid ratio sludges and kinetic analysis. RSC Adv. 2019;9:19104–13. https://doi.org/10.1039/c9ra01662g.
123. Tasca AL, et al. Hydrothermal carbonization of sewage sludge: A critical analysis of process severity, hydrochar properties and environmental implications. Waste Manag. 2019;93:1–13. https://doi.org/10.1016/j.wasman.2019.05.027.
124. Wei Y, et al. Hydrothermal liquefaction of municipal sludge and its products applications. Sci Total Environ. 2024;908:168177. https://doi.org/10.1016/j.scitotenv.2023.168177.
125. Thant Zin MM, Kim DJ. Simultaneous recovery of phosphorus and nitrogen from sewage sludge Ash and food wastewater as Struvite by Mg-biochar. J Hazard Mater. 2021;403:123704. https://doi.org/10.1016/j.jhazmat.2020.123704.
126. Günther S, Grunert M, Müller S. Overview of recent advances in phosphorus recovery for fertilizer production. Eng Life Sci. 2018;18:434–9. https://doi.org/10.1002/elsc.201700171.
127. Cifuentes-Cabezas M, et al. Nitrogen recovery from sludge centrate by membrane contactor: influence of operating parameters and cleaning conditions. J Environ Manage. 2023;341:118051. https://doi.org/10.1016/j.jenvman.2023.118051.
128. Pagliano G, et al. Recovery of polyhydroxyalkanoates from single and mixed microbial cultures: A review. Front Bioeng Biotechnol. 2021;9:1–28. https://doi.org/10.3389/fbioe.2021.624021.
129. Conca V, et al. Long-term validation of polyhydroxyalkanoates production potential from the sidestream of municipal wastewater treatment plant at pilot scale. Chem Eng J. 2020;390:124627. https://doi.org/10.1016/j.cej.2020.124627.
130. Corsino SF, et al. Integrated production of biopolymers with industrial wastewater treatment: effects of OLR on process yields, biopolymers characteristics and mixed microbial community enrichment. J Water Process Eng. 2022;47:102772. https://doi.org/10.1016/j.jwpe.2022.102772.

131. Guo J, Ma J. Bioflocculant from pre-treated sludge and its applications in sludge dewatering and swine wastewater pretreatment. Bioresour Technol. 2015;196:736–40. https://doi.org/10.1016/j.biortech.2015.07.113.
132. Schambeck CM, et al. Chemical and physical properties of alginate-like exopolymers of aerobic granules and Flocs produced from different wastewaters. Bioresour Technol. 2020;318:123632. https://doi.org/10.1016/j.biortech.2020.123632.
133. Taghvaie Nakhjiri A, et al. Recovery of precious metals from industrial wastewater towards resource recovery and environmental sustainability: A critical review. Desalination. 2022;527:115510. https://doi.org/10.1016/j.desal.2021.115510.
134. Yesil H, et al. Removal and recovery of heavy metals from sewage sludge via three-stage integrated process. Chemosphere. 2021;280:130650. https://doi.org/10.1016/j.chemosphere.2021.130650.
135. Chang Z, et al. Valorization of sewage sludge in the fabrication of construction and Building materials: A review. Resour Conserv Recycl. 2020;154:104606. https://doi.org/10.1016/j.resconrec.2019.104606.
136. Souza M, et al. Developing and classifying lightweight aggregates from sewage sludge and rice husk Ash. Case Stud Constr Mater. 2020;12:e00340. https://doi.org/10.1016/j.cscm.2020.e00340.
137. Xu Y, et al. Phosphorus recovery from sewage sludge Ash (SSA): an integrated technical, environmental and economic assessment of wet-chemical and thermochemical methods. J Environ Manage. 2023;344:118691. https://doi.org/10.1016/j.jenvman.2023.118691.
138. Franus M, et al. Utilization of sewage sludge in the manufacture of lightweight aggregate. Environ Monit Assess. 2016;188(10). https://doi.org/10.1007/s10661-015-5010-8.
139. Bagheri M, et al. Fifty years of sewage sludge management research: mapping researchers' motivations and concerns. J Environ Manag. 2023;325:116412.
140. Zhuang Z, et al. An economic and global warming impact assessment of common sewage sludge treatment processes in North America. J Clean Prod. 2022;370:133539.
141. Arias A, Feijoo G, Moreira MT. Benchmarking environmental and economic indicators of sludge management alternatives aimed at enhanced energy efficiency and nutrient recovery. J Environ Manag. 2021;279:111594.
142. Raheem A, et al. Opportunities and challenges in sustainable treatment and resource reuse of sewage sludge: A review. Chem Eng J. 2018;337:616–41. https://doi.org/10.1016/j.cej.2017.12.149.
143. Medina-Martos E, et al. Techno-economic and life cycle assessment of an integrated hydrothermal carbonization system for sewage sludge. J Clean Prod. 2020;277:122930.
144. Xu C, Chen W, Hong J. Life-cycle environmental and economic assessment of sewage sludge treatment in China. J Clean Prod. 2014;67:79–87.
145. Li H, et al. Environmental and economic life cycle assessment of energy recovery from sewage sludge through different anaerobic digestion pathways. Energy. 2017;126:649–57. https://doi.org/10.1016/j.energy.2017.03.068.
146. Huang H, et al. Persulfate oxidation for alternative sludge treatment and nutrient recovery: an assessment of technical and economic feasibility. J Environ Manage. 2020;272:111007. https://doi.org/10.1016/j.jenvman.2020.111007.
147. Garbowski T, et al. An overview of natural soil amendments in agriculture. Soil Tillage Res. 2023;225:105462. https://doi.org/10.1016/j.still.2022.105462.
148. Eden M, Gerke HH, Houot S. Organic waste recycling in agriculture and related effects on soil water retention and plant available water: A review. Agron Sustain Dev. 2017;37(2):11. https://doi.org/10.1007/s13593-017-0423-1.
149. Obriot F, et al. Multi-criteria indices to evaluate the effects of repeated organic amendment applications on soil and crop quality. Agric Ecosyst Environ. 2016;232:165–78. https://doi.org/10.1016/j.agee.2016.08.006.
150. Bondì C, Castellini M, Iovino M. Compost amendment impact on soil physical quality estimated from hysteretic water retention curve. Water. 2022;14(7):1090. https://doi.org/10.3390/w14071090.
151. Głąb T, et al. Effects of co-composted maize, sewage sludge, and Biochar mixtures on hydrological and physical qualities of sandy soil. Geoderma. 2018;315:27–35. https://doi.org/10.1016/j.geoderma.2017.11.030.
152. Zuo W, et al. Sewage sludge amendment improved soil properties and sweet sorghum yield and quality in a newly reclaimed mudflat land. Sci Total Environ. 2019;654:541–9. https://doi.org/10.1016/j.scitotenv.2018.11.127.
153. Chen H, et al. An 18-year field experiment to assess how various types of organic waste used at European regulatory rates sustain crop yields and C, N, P, and K dynamics in a French calcareous soil. Soil Tillage Res. 2022;221:105415. https://doi.org/10.1016/j.still.2022.105415.
154. Bondì C, et al. Cactus Pear pruning residue in agriculture: unveiling soil-specific responses to enhance water retention. Environ Technol Innov. 2024;34:103602. https://doi.org/10.1016/j.eti.2023.103602.
155. Ma D, Wang Y, Ye Y. Effects of three sludge products from co-treatment of wastewater on the soil properties and plant growth of silty loam. Int J Environ Res Public Health. 2022;19(7):4146. https://doi.org/10.3390/ijerph19074146.
156. Soria R, et al. Role of organic amendment application on soil quality, functionality and greenhouse emission in a limestone quarry from semiarid ecosystems. Appl Soil Ecol. 2021;164:103925. https://doi.org/10.1016/j.apsoil.2021.103925.
157. Simões-Mota A, Virto I, Poch RM. Effects of long-term sewage sludge application to a calcareous soil structure. Soil Use Manag. 2022;38:1693–704. https://doi.org/10.1111/sum.12891.
158. Yameen MZ, et al. Harnessing the power of functionalized biochar: progress, challenges, and future perspectives in energy, water treatment, and environmental sustainability. Biochar. 2024;6(1):25. https://doi.org/10.1007/s42773-024-00316-3.
159. Singh RP, Agrawal M. Potential benefits and risks of land application of sewage sludge. Waste Manag. 2008;28:347–58. https://doi.org/10.1016/j.wasman.2006.12.010.
160. Shan Y, et al. Sewage sludge application enhances soil properties and rice growth in a salt-affected mudflat soil. Sci Rep. 2021;11:1402. https://doi.org/10.1038/s41598-021-80395-5.
161. Dhanker R, et al. Influence of urban sewage sludge amendment on agricultural soil parameters. Environ Technol Innov. 2021;23:101642. https://doi.org/10.1016/j.eti.2021.101642.
162. Eid EM, et al. Sewage sludge enhances tomato growth and improves fruit-yield quality by restoring soil fertility. Plant Soil Environ. 2021;67(9):514–23. https://doi.org/10.17221/90/2021-PSE.
163. Angin I, Yaghanoglu AV. Effects of sewage sludge application on some physical and chemical properties of a soil affected by wind erosion. J Agric Sci Technol. 2011;13(5):757–68.
164. Laudicina VA et al. Key biochemical attributes to assess soil ecosystem sustainability. Environ Prot Strategies Sustain Dev, 193–227 (2012).

165. Zoghlami RI, et al. Changes in light-textured soil parameters following two successive annual amendments with urban sewage sludge. Ecol Eng. 2016;95:604–11. https://doi.org/10.1016/j.ecoleng.2016.06.054.
166. Mondal S, et al. Changes in soil quality in response to short-term application of municipal sewage sludge in a typic haplustept under cowpea-wheat cropping system. Environ Nanotechnol Monit Manage. 2015;4:37–41. https://doi.org/10.1016/j.enmm.2015.06.002.
167. Skowrońska M, et al. An integrated assessment of the long-term impact of municipal sewage sludge on the chemical and biological properties of soil. CATENA. 2020;189:104484. https://doi.org/10.1016/j.catena.2020.104484.
168. Gianico A, et al. Land application of biosolids in europe: possibilities, constraints and future perspectives. Water. 2021;13(1):103. https://doi.org/10.3390/w13010103.
169. Száková J, et al. Selected persistent organic pollutants (POPs) in the rhizosphere of sewage sludge-treated soil: implications for the biodegradability of pops. Arch Agron Soil Sci. 2019;65:994–1009. https://doi.org/10.1080/03650340.2018.1513512.
170. Aydın S, et al. Pharmaceutical residues in digested sewage sludge: occurrence, seasonal variation and risk assessment for soil. Sci Total Environ. 2022;817:152864. https://doi.org/10.1016/j.scitotenv.2022.152864.
171. Latosińska J, Kowalik R, Gawdzik J. Risk assessment of soil contamination with heavy metals from municipal sewage sludge. Appl Sci. 2021;11(2):548. https://doi.org/10.3390/app11020548.
172. Wang J, et al. Mechanisms of soil N dynamics following long-term application of organic fertilizers to subtropical rain-fed purple soil in China. Soil Biol Biochem. 2015;91:222–31. https://doi.org/10.1016/j.soilbio.2015.08.023.
173. Li L, et al. Factors affecting soil microbial biomass and functional diversity with the application of organic amendments in three contrasting cropland soils during a field experiment. PLoS ONE. 2018;13(9):e0203812. https://doi.org/10.1371/journal.pone.0203812.
174. Paliaga S, et al. Long-term organic management: mitigating land use intensity drawbacks and enhancing soil microbial redundancy. J Plant Nutr Soil Sci. 2024;187:287–94. https://doi.org/10.1002/jpln.202300252.
175. Charlton A, et al. Long-term impact of sewage sludge application on soil microbial biomass: an evaluation using meta-analysis. Environ Pollut. 2016;219:1021–35. https://doi.org/10.1016/j.envpol.2016.07.050.
176. Paz-Ferreiro J, et al. Soil biochemical activities and the geometric mean of enzyme activities after application of sewage sludge and sewage sludge Biochar to soil. Biol Fert Soils. 2012;48:511–7. https://doi.org/10.1007/s00374-011-0657-3.
177. Rehana MR, Joseph B, Gladis R. Heavy metal stabilization in sewage sludge composting process. Curr J Appl Sci Technol. 2020;39(19):38–48. https://doi.org/10.9734/cjast/2020/v39i1930836.
178. Lü H, et al. Occurrence and dissipation mechanism of organic pollutants during the composting of sewage sludge: A critical review. Bioresour Technol. 2021;328:124847. https://doi.org/10.1016/j.biortech.2021.124847.
179. Curci M, et al. Short-term effects of sewage sludge compost amendment on semiarid soil. Soil Syst. 2020;4(3):48. https://doi.org/10.3390/soilsystems4030048.
180. Adetunji AT, et al. The biological activities of β-glucosidase, phosphatase and urease as soil quality indicators: A review. J Soil Sci Plant Nutr. 2017;17(3):794–807. https://doi.org/10.4067/S0718-95162017000300024.
181. Kandeler E, et al. Structure and function of the soil microbial community in microhabitats of a heavy metal polluted soil. Biol Fert Soils. 2000;32:390–400. https://doi.org/10.1007/s003740000251.
182. Arif MS, et al. Fresh and composted industrial sludge restore soil functions in surface soil of degraded agricultural land. Sci Total Environ. 2018;619:517–27. https://doi.org/10.1016/j.scitotenv.2017.11.136.
183. Kızılkaya R, Bayraklı B. Effects of N-enriched sewage sludge on soil enzyme activities. Appl Soil Ecol. 2005;30(3):192–202. https://doi.org/10.1016/j.apsoil.2005.03.003.
184. Li Y, et al. Safety and efficiency of sewage sludge and garden waste compost as a soil amendment based on the field application in woodland. Ecotoxicol Environ Saf. 2021;222:112497. https://doi.org/10.1016/j.ecoenv.2021.112497.
185. Melo W, et al. Ten years of application of sewage sludge on tropical soil: A balance sheet on agricultural crops and environmental quality. Sci Total Environ. 2018;643:1493–501. https://doi.org/10.1016/j.scitotenv.2018.06.135.
186. Sciubba L, et al. Relationships between stability, maturity, water-extractable organic matter of municipal sewage sludge composts and soil functionality. Environ Sci Pollut Res. 2015;22:13393–403. https://doi.org/10.1007/s11356-015-4526-9.
187. Hechmi S, et al. Carbon mineralization, biological indicators, and phytotoxicity to assess the impact of urban sewage sludge on two light-textured soils in a microcosm. Environ Toxicol Chem. 2020;49(2):460–71. https://doi.org/10.1002/etc.4929.
188. Zhao Q, et al. Sewage sludge application alters the composition and co-occurrence pattern of the soil bacterial community in Southern China forestlands. Appl Soil Ecol. 2021;157:103744. https://doi.org/10.1016/j.apsoil.2020.103744.
189. Hao J, et al. The effects of soil depth on the structure of microbial communities in agricultural soils in Iowa (United States). Appl Environ Microbiol. 2021;87(4):e02673–20. https://doi.org/10.1128/AEM.02673-20.
190. Bastida F, et al. Application of fresh and composted organic wastes modifies structure, size and activity of soil microbial community under semiarid climate. Appl Soil Ecol. 2008;40:318–29. https://doi.org/10.1016/j.apsoil.2008.01.003.
191. Yin T, et al. Impact of soil physicochemical factors and heavy metals on co-occurrence pattern of bacterial in rural simple garbage dumping site. Ecotoxicol Environ Saf. 2024;280:116476. https://doi.org/10.1016/j.ecoenv.2024.116476.
192. Dahiya P, Singh N, Singh A. Impact of sewage sludge application on microbial diversity and fertility of soil: A long-term study. In *Development in Waste Water Treatment Research and Processes*. https://doi.org/10.1016/B978-0-323-85584-6.00006-6 (2022).
193. Cheng H, et al. An assessment of soil microbial community environmental stress in alpine marginal ecosystems using microbial indicators. Ecol Indic. 2024;158:111542. https://doi.org/10.1016/j.ecolind.2023.111542.
194. Hua Y, et al. Changes in soil microbial community structure following amendment of biosolids for seven years. Environ Pollut Bioavailab. 2019;31(1):24–31. https://doi.org/10.1080/26395940.2019.1569478.
195. You J, et al. Effects of sewage sludge Biochar on soil characteristics and crop yield in loamy sand soil. Pol J Environ Stud. 2019;28(4):2973–80. https://doi.org/10.15244/pjoes/93294.

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